CARABID BEETLES AS BIODIVERSITY AND ECOLOGICAL INDICATORS

By

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Catadromus lacordairei (Boisduval)

Carabids are beeetles of ground
So spots where carabids are found
Is reason for inferring
That there they're occurring
This circular reasoning is round

Without wings they're more apt to stay there
But the winged may take to the air
Dispersing in myriads
Through Tertiary periods
We know they all started, but *where*?

Erwin *et al.* 1979

DECLARATION

This thesis contains no material which has been accepted for the award of any other degree or diploma in any tertiary institution and to the best of my knowledge and belief, the thesis contains no material previously published or written by another person, except when due reference is made in the text.

Karyl Michaels

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STATEMENT OF PREVIOUS PUBLICATONS

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ABSTRACT

The explicit assumption in the proposed use of ecological and biodiversity indicators is that patterns of diversity and distribution observed in the indicator taxon are reflected in other taxa, yet this assumption has rarely been tested. The use of biodiversity and ecological indicators requires that their representativeness of other taxa be demonstrated. This thesis examines the potential for using carabid beetles as biodiversity and ecological indicators for other Coleoptera in Tasmania.

Species-occurrence data for carabids and a range of Coleoptera collected by continuous year-long pitfall trapping at fifty-one sites in three biomes, dry sclerophyll forest, remnant native grasslands and coastal sand dunes were used to investigate the utility of carabid beetles as biodiversity indicators for overall coleopteran diversity and for other selected beetle taxa.

Correlated species counts, correlations in rank ordering based on species richness, and coincidence of hotspots revealed that while the patterns of diversity exhibited by carabids did not necessarily predict the patterns of diversity of other individual beetle taxa considered separately, they did, with a reasonable degree of accuracy, indicate overall patterns of diversity for Coleoptera in all the biomes studied. Carabid species richness was a good predictor of overall beetle species richness within biomes and within vegetation community types.

Application of three reserve selection approaches: (1) Hotspots, (2) Representative Species Richness and (3) Complementarity, demonstrated that a set of representative areas, based on carabids species richness, gave proportional representation for all Coleoptera. Representation for all carabids using the complementarity approach also gave protection to over 90% of all coleopteran species. It is therefore likely that a set of representative areas in which carabids are completely represented will substantially represent the diversity of other Coleoptera.

To assess the utility of carabids as ecological indicators for other Coleoptera, the response of carabids and other Coleoptera to silvicultural practices (clearfelling and slash burning) were examined and compared.

Morisita-Horn community similarity indices demonstrate that carabid and overall Coleoptera species composition showed less variation within grouped age classes than between different age classes in both forests. Results for other beetle taxa

considered separately were more complex and varied with forest. In both forests, UPGMA cluster analysis generally grouped the total beetle fauna according to regrowth age, but indicated that the species composition of regrowth sites were often similar to some old-growth sites. This pattern was observed for carabids in the dry sclerophyll forest, but not in the wet. Other beetle taxa demonstrated more complex patterns of clustering, with no clear evidence of site separation on age class.

The family Carabidae did not reflect the exact response of other beetle families considered separately. However, they did reflect the overall patterns of diversity and distribution exhibited by beetles as a group in response to forest management practices. Results demonstrated that monitoring particular carabid species would provide evidence of the success or otherwise of management practices for other old-growth dependent beetles.

The results reported in this thesis support the hypotheses that:

(i) the family of ground beetles (Carabidae) is an appropriate biodiversity indicator for identifying and predicting the biodiversity patterns of ground dwelling Coleoptera in most instances in Tasmania, and

(ii) that carabids are useful ecological indicators to predict and monitor the effects of forest management on a wider range of ground-dwelling beetles in Tasmania.

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Chapter One

Introduction

This thesis addresses the complex issue of the use of biodiversity and ecological indicators in the domain of conservation biology. The explicit assumption in the promotion of ecological and biodiversity indicators is that patterns of diversity and distribution observed in the indicator taxon are reflected in other taxa, yet this assumption has rarely been tested. The effective use of biodiversity and ecological indicators requires that their representativeness of other taxa be demonstrated. This thesis attempts such a demonstration by testing the potential for using carabid beetles as biodiversity and ecological indicators for other Coleoptera in Tasmania.

BIODIVERSITY IS BEETLES

Asked what could be concluded about the nature of the creator from a study of his creation, the noted biologist J. B. S. Haldane is alleged to have replied "An inordinate fondness for beetles" (Gould 1993 p. 4.). Beetles are a substantial part of the global fauna and of the total global biodiversity (Fig. 1.1). Beetles epitomise biodiversity.

Over half of all known animal species are insects (New *et al.* 1995), and estimates of insect diversity range from 5 million to as many as 80 million species (Gullan and Cranston 1994). Measured by species, individuals or biomass, insects are the most diverse components of terrestrial ecosystems (Wilson 1987; Stork 1988; Erwin 1991; Kremen *et al.* 1993). The largest group of insects and the largest Order in the Animal Kingdom is the Coleoptera (beetles) (Evans and Bellamy 1996). Beetles represent one quarter of all known animal species on earth and one fifth of all known living organisms (Evans and Bellamy 1996). If insects are "the little things that run the world" (Wilson 1987), beetles are the frontline troops. Beetles are tiny recycling machines, recycled in turn by other animals and plants. Practically every biological strategy used by terrestrial insects is employed by beetles.

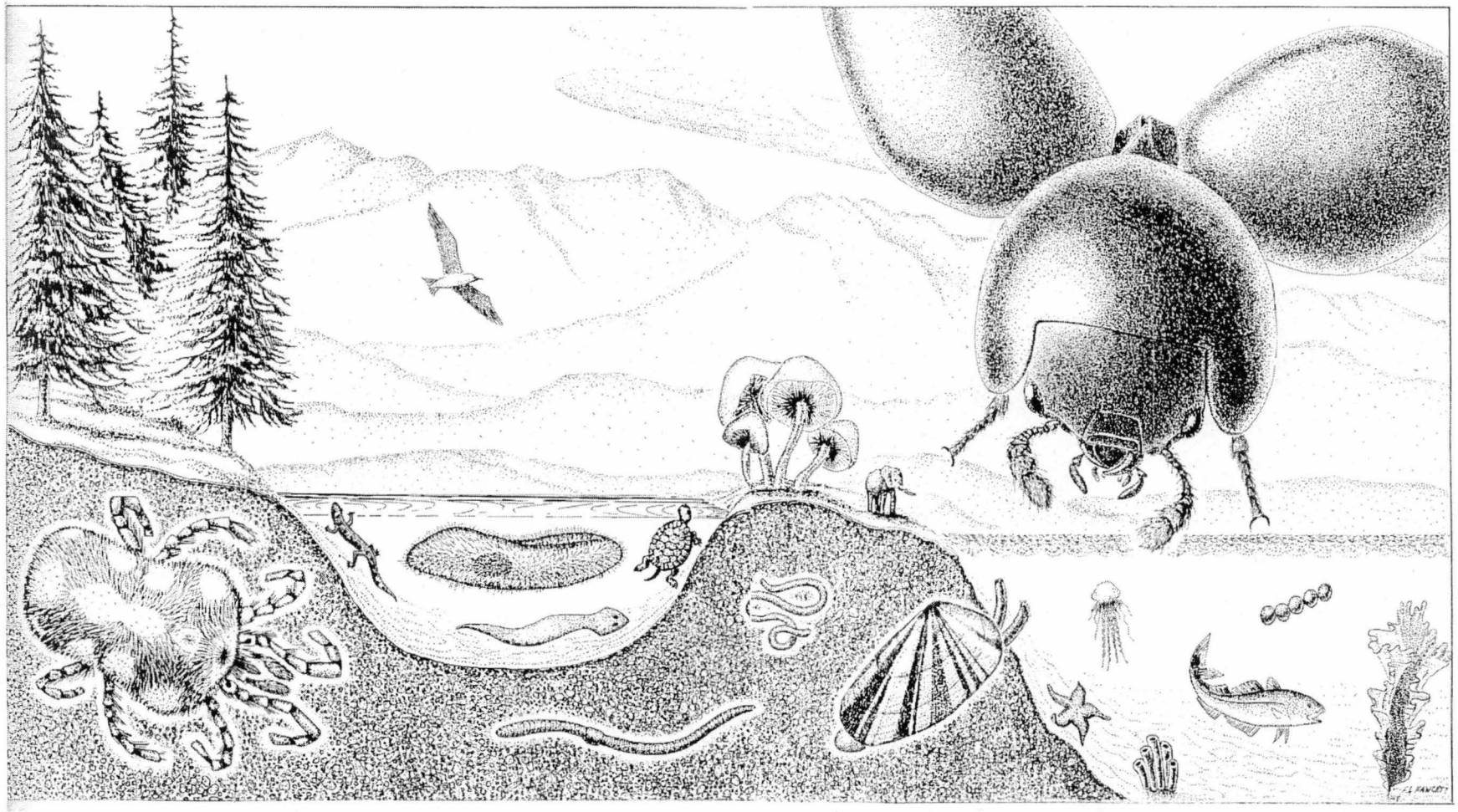

Figure 1.1. Wheeler's speciescape, on which size of individual organisms approximately corresponds to number of described species in higher taxon that it represents. From Wheeler 1990, original by Frances Fawcett. The *Agathidium oniscoides* (Coleoptera: Leiodidae) in flight represents currently named insects, around 750,000 species.

Their hard cuticle and tough elytra allow them to exploit almost every conceivable habitat: from the depths of subterranean caverns to mountain tops; from arid deserts to rainforests; from intertidal shorelines to riparian zones; every vegetational microhabitat from external foliage flowers, buds, stems barks and roots to internal sites such as galls in living plant tissue; and any kind of dead material in all various states of decomposition (Britton 1991; Gullan and Cranston 1994; Evans and Bellamy 1996).

BIODIVERSITY IS IN CRISIS

But, biodiversity is in decline on an unprecedented scale (Kirkpatrick *et al.* 1995; Pimm and Lawton 1998). Biodiversity, short for biological diversity, encompasses the variety of earth's biological resources (McFadgen and Simpson 1994). The protection and conservation of biodiversity has never been more urgent, with species, communities and ecosystems disappearing at an alarming and ever increasing rate (Ehrlich and Wilson 1991).

Man himself, has been and continues to be, the biggest threat to biodiversity, with our rapacious pursuit of a higher standard of living for an ever growing human population. Habitat loss, destruction and degradation through land clearance, wetland reclamation, coastal, urban and industrial developments and mining; conversion of much of the environment to agricultural and forestry use accompanied by the fragmentation of remaining habitat is the major cause of species loss (DEST 1995). The exploitation of our biological resources as if they were boundless has resulted in dramatic changes to habitats and ecosystems world-wide (Paoletti 1995).

CONSERVATION AND SUSTAINABILITY OF BIODIVERSITY

Conservation biology is concerned with the conservation and sustainable management of biodiversity and has two primary objectives:

(1) to represent the greatest possible diversity (of living organisms) in a reserve system and to ensure their survival (Margules and Nicholls 1987), and

(2) to understand and control human impacts sufficiently to ensure the sustainable survival of all the other species that share the world (New 1995).

To achieve the first objective we need to:

(1) identify where the greatest diversity is located and identify a set of sites representative of overall diversity,

(2) identify best management and planning strategies to ensure its continued existence and

(3) monitor for sustainability to ensure its continued existence

To achieve the second objective, we need to:

(1) evaluate, assess and understand the effects of differing land management practices

(2) identify best management practices and

(3) identify and recognise ecological change due to land management practices before it becomes critical and irreversible.

Conservation and sustainability therefore, have two main elements: inventory and monitoring.

Inventory - the spatial quantification of biodiversity

The purpose of inventory programs is to document the spatial distribution of biological elements - species, populations, communities and ecosystems (Kremen *et al.* 1993). Knowledge of species distribution and habitat requirements is vital for identification of critical habitat, ecological evaluation, reserve design and selection (New 1995). Without data on diversity, both the amount and the distribution we have no way of knowing what levels of diversity are present and therefore what levels of diversity may be lost if not protected (Hawkes 1992). An adequate inventory of species is required for both conservation and sustainable use of species because you need to know what you have before you can make decisions about sustainable resource use or protection (Prance 1995).

Information on diversity is not only vital for the design of conservation schemes, but is also essential for assessing the environmental impacts of human caused disturbances (Hawkes 1992; New 1995). Inventories provide quantitative data from which comparisons can be made based on qualitative descriptions with existing classifications of ecosystems (on the basis of any number of characteristics). This can be used in planning a national system of reserves which

will conserve the range of variability within and between regions (Pressey *et al.* 1993). But, evaluating and understanding the results of land use practices, identifying best management practices and recognising ecological changes also requires baseline data. We need to know what we have if we are to recognise and understand the impacts of our actions and modify them.

Monitoring - the spatial and temporal quantification of biodiversity

Monitoring is the quantification of biodiversity over time as well as space, and is essential for demonstrating long term sustainability. If we are to ensure survival of extant species, we need to monitor to detect early changes or stresses to natural systems (New 1995). If we are to successfully change and/or modify our actions, we need to monitor to assess the success or otherwise of our management practices (McLaren *et al.* 1998). Monitoring programmes should ideally provide a sensitive and accurate source of information from which conditions and trends can be defined and recognised, and management decisions made (Thomas 1993).

Land use is expanding as is land degradation. While some forms of land degradation (i.e. soil erosion, salination, extensive overgrazing, desertification) are distinct and highly visible, the effects of most land management are subtle and their ecological consequences may not be immediately apparent (Andersen 1990). There is an urgent need to assess what effects variations in land use have on resident species, and it is important to be able to recognise ecological change well before it becomes dramatic and possibly irreversible (Andersen 1990). Monitoring of selected species can identify changes in biological diversity, permitting the timely adjustment of management activities to reverse or avoid undesired trends (Sparrow *et al.* 1994). Monitoring tells us how to best modify land use practices to both maximise diversity and protect resident fauna for the long term (that is to prevent species becoming extinct, threatened or rare). Hopefully, monitoring will warn us if we are doing the wrong thing before we get to the point of no return.

INDICATORS

Ideally, we should inventory and monitor all biodiversity. Such baseline data are a "vitally needed prerequisite for rational planning and for the preservation of biological diversity" (Soulé and Kohm 1989; Hawkes 1992; New 1995). But, unfortunately, there is too little time, too little money or resources, too little expertise, and too much diversity (Vane-Wright *et al.* 1994; Balmford *et al.* 1996a. Yet, we need reliable data to inform debate on how best to counter threats and

implement appropriate conservation measures (Hammond 1994). The biodiversity crisis, the severity and expanding nature of the problem, and the inherent difficulties in documenting and monitoring biodiversity, has therefore led to increasing interest in the concept of indicators.

Indicators are organisms (taxa or suites of taxa) whose key attributes or characteristics are measured or monitored under the assumption that they reflect the condition and/or trend of some ecological property that is too difficult, inconvenient or expensive to measure for other species, or to monitor directly (Landres *et al.* 1988; Noss 1990).

McGeoch (1998) suggests that there are three categories of indicators corresponding to their main applications.

- environmental indicators - species or taxa that demonstrate a predictable response to environmental disturbance or change.

- biodiversity indicators - species or taxa whose diversity reflects some measure of the diversity of other taxa.

- ecological indicators - species or taxa that demonstrate the effects of environmental disturbance or change on at least a subset of other taxa

Environmental indication is the traditional use of indication, where an indicator taxon is monitored so that changes in environmental condition can be detected. Canaries down the mines are probably one of the earliest examples of a species used as an indicator of environmental quality. Aquatic invertebrates have been used extensively as indicators for environmental impact assessment in both freshwater and marine systems for more than 30 years (Hellawell 1986; Rosenberg and Resh 1993).

However, as anthropogenic impacts on the environment have increased, and the need to protect and manage natural resources has become increasingly critical, the potential applications for indicators have expanded to meet current needs. Biodiversity indicators have been proposed to obviate the need for total species inventories, and ecological indicators have been proposed to obviate the need for monitoring the impacts of anthropogenic activities on all biota.

TESTING THE INDICATOR CONCEPT

Legitimate use of biodiversity and ecological indicators require that they are an adequate surrogate for other biota. Biodiversity indicators should accurately represent aspects of diversity of other organisms and conservation based on biodiversity indicators should ensure protection for uninventoried taxa via the precautionary principle. Ecological indicators should accurately reflect the impacts of environmental changes or stresses on other unmonitored organisms.

Many species have been suggested as indicator species. Many authors have suggested criteria to be applied to assess the suitability or potential of taxa as indicators. However, although the uses, merits and demerits of indicators have been much discussed, and there has been extensive discussion on the potential use of indicators, their representativeness of other taxa has been little tested. The use of biodiversity and ecological indicators requires that their representativeness of other taxa be demonstrated.

CARABIDS AS INDICATORS

The family Carabidae (ground beetles) includes over 40, 000 species globally. It represents 3% of all insects, more species than almost any other insect family and 2.5% of all animal species (Thiele 1977). Carabids are one of the largest beetle families and numerous studies suggest that they fill the *a priori* criteria to be an indicator taxon (for example, Thiele 1977; Stork 1990; Desender *et al.* 1991b).

Carabids are represented on a more or less global basis, and present in almost all kinds of habitats, generally in reasonable numbers, all the year round (Thiele 1977). Most species are sensitive to environmental changes in their habitat with ecological requirements limited enough either to allow useful predictions to be made about changes in quality and land use history, or to characterise the community and the habitat (Stork 1990; Lövei and Sunderland 1996). Like Coleoptera generally, they have adapted to fill a range of niches. Collectively, they are predatory, omnivorous, even phytophagous with a taste for strawberries and they are considered potentially important natural pest control agents in agroecosystems (Clark *et al.* 1997). They are easily and cost-effectively sampled with modest effort and without the use of specialised equipment (Spence and Niemelä 1994). Readily identifiable, they are one of the most widely studied taxa in the world.

It would be advantageous if Carabids were indicators for general coleopteran diversity. Then we could inventory carabids and be assured that other Coleoptera were being considered in biodiversity assessments and strategies. It would also be advantageous if carabids were ecological indicators for other Coleoptera. If we were assured that the responses of beetles as a group mirrored the responses exhibited by carabids, then we could monitor carabids and we would know the impact of an environmental stressor on a large part of the fauna.

The utility of carabids as environmental indicators has been relatively well documented in other regions. But their usefulness as biodiversity indicators and ecological indicators has yet to be tested in Australia.

THESIS AIMS AND STRUCTURE

The aims of this thesis are:

- to test the hypothesis of utility of carabid beetles as a biodiversity indicator for other taxa: specifically, other selected Coleoptera.

- to test the hypothesis of utility of carabid beetles as an ecological indicator for other taxa: specifically, other selected Coleoptera.

A range of sites were surveyed and inventoried

As a result of these surveys, several research hypotheses were tested:

1. Carabid diversity accurately indicates/reflects general beetle biodiversity.

2. Reserve selection based on carabids will provide protection for other selected Coleoptera.

3. The response of carabids to a major environmental stressor (clearfell logging) is representative of the response of other selected Coleoptera.

The traditional and current definitions of indicators will be reviewed, practical considerations for using indicators will be discussed, and the criteria for selecting indicator species will be examined in **Chapter Two**. A number of desirable and necessary characteristics of indicators are repeatedly suggested. The extent to

which carabid beetles meet these criteria will be addressed in **Chapter Three.** The data used in this thesis was obtained from several different studies undertaken at a range of locations in Tasmania. **Chapter Four** describes the location, vegetation and environmental parameters of the study sites. Sites were inventoried to produce species lists of carabids and other Coleoptera. The sampling and identification protocols used to produce these lists are described and discussed in **Chapter Five.**

The hypothesis that the family of ground beetles (Carabidae) is an appropriate biodiversity indicator taxon for identifying and predicting the biodiversity patterns of other selected Coleoptera is examined in **Chapter Six.** Common test measures of indicators from the literature will be used to test whether patterns of diversity of carabids are congruent with the diversity patterns of other selected Coleoptera and different reserve selection approaches will be applied to assess the extent to protection for carabids might result in the conservation of Coleopteran biodiversity.

In **Chapter Seven** the hypothesis that carabids are useful ecological indicators to predict and monitor the effects of forest management activities on a wider range of ground-dwelling beetles will be examined. The response of carabids and other selected Coleoptera to commercial forestry harvest and regeneration practices (clearfelling followed by burning and regeneration) will be evaluated and compared to assess if the responses of carabids to this environmental disturbance are representative of the response of other Coleoptera.

The final chapter (**Chapter Eight**) is a synthesis that brings together the conclusions which could be drawn regarding the thesis aims and the practical considerations for using carabids as biodiversity and ecological indicators will be discussed.

Chapter Two

Indicators

Indicator species have been used for decades as a convenient assay of environmental conditions (de Wit 1983; Eijsackers 1983; Hellawell 1986; Rosenberg and Resh 1993), and the characterisation of habitats by species assemblages (indicator species) is a core activity in ecology and biogeography (Dufrene and Legendre 1997).

The use of indicators is receiving increased interest in conservation biology, indeed, current conservation strategies strongly mandate for their use. However, the conceptual bases, the assumptions and criteria for the use of indicator species are as varied as the proposed uses. In this chapter, I: (1) review traditional and current definitions of indicators; (2) discuss the issues involved in using indicators; and (3) examine the criteria for selecting indicator species.

WHAT IS AN INDICATOR

"Indicate" from the Latin *indicare* to point out, to show (Hanks 1979 p745).

The Collins Dictionary of Biology defines an indicator species as "any species that is indicative of particular conditions or habitats" and an indicator community as "an association of species that is characteristic of particular conditions or habitats" (Hale and Margham 1988 p 292) .

The commonplace observation that organisms reflect their environment is found in writings dating from as early as Aristotle (Cranston *et al.* 1996). The indicator concept itself is as old as man's first attempt to deal with the variables in his environment by organising his qualitative observations into a cause and effect system of prediction (Soule 1988). For example, farmers and shepherds recognised that the presence or growth of particular plants indicated particular soil qualities and used this knowledge to choose fields and pastures (Zonneveld 1983). By feeding them part of the catch, the natives of some tropical islands use the response of cats to indicate the presence of the toxic dinoflagellates which cause lethal *Ciguatera* poisoning in reef fish, leading to a short supply of cats in some

instances (Soule 1988). Fishermen used (and still do despite the advent of SONAR) the presence of circling birds as a visual indicator of the presence of schooling fish.

A literature search indicates that there has been a big increase in publications on the subject of indicators in recent years. One of the major factors driving this interest in indicators is increased concerns about how to conserve and sustain biodiversity in the face of accelerating habitat loss, destruction and degradation. The urgent need to assess the quality of ecosystems, to identify key components which can serve as indicators of ecosystem quality and to detect and manage ecosystem changes following perturbations has become a major issue globally and the increased interest in indicators reflects this.

Indeed, increasing concern for the environmental impact of human activities has led to many countries mandating for the use of indicators to assess, monitor and document biological diversity and environmental quality. Australia along with 156 other countries is signatory to the (1992) Convention on Biological Diversity. The convention calls upon its member states to identify components of biological diversity important for the long-term conservation and sustainability of biodiversity such as indicator species (McFadgen and Simpson 1994; McGeoch 1998). It also calls upon signatories to monitor the components of biological diversity and identify activities which are likely to have significant adverse impacts. In order to meet its international obligations, Australia intends to develop a "core set "of biodiversity indicators to assess and monitor the condition of Australia's biological diversity (Saunders *et al.* 1998).

The term indicators is being employed in a wide range of contexts, including the indication of sustainable development, ecosystem health, habitat alteration, rehabilitation, pollution, climate change and even past climates (Hunsaker and Carpenter 1990; New 1995; Bertollo 1998; Whitford *et al.* 1998). There has also been a proliferation of related terms and synonyms (see for example Spellerberg 1991; Hammond 1994; Vane-Wright *et al.* 1994; New 1995; Cranston and Trueman 1997; Fisher 1998; McGeoch 1998). Indicators are also referred to as bioindicators (biological indicators), the term originating in the aquatic literature where it referred to the detection and monitoring of change in biota to reflect changes in the environment (Wilhm and Dorris 1968). The term indicator will be used hereafter for convenience. The definition of indicators is thus largely dependent on their purpose or application.

Types of indicators

McGeoch (1998) suggests that there are three categories or types of indicators corresponding to the three main applications of indicators: environmental indicators, ecological indicators and biodiversity indicators.

(1) Environmental indicators

Environmental indication is the traditional use of indication, where an indicator taxon is monitored so that changes in environmental condition can be detected (McGeoch 1998). Environmental indicators are individual species (taxon) or species assemblages (taxa) that demonstrate a readily observable, quantifiable and predictable response to disturbance or change in environmental condition (McGeoch 1998) (one environmental condition being habitat type). Odum 1971) remarked that the environment of organisms can be judged on the basis of the species that are present: the indicator species. For example, chironomid midge community structure is considered the most important indicator of lake typology in the northern hemisphere (Cranston 1990).

Environmental indicators incorporate both the condition or state of the environment (biophysical indicators) and stresses on the environment by anthropogenic activities (e.g. toxic emissions, energy use, natural resource harvesting (Bertollo 1998). By expressing particular symptoms or responses indicative of changes in some environmental influence, usually in a qualitative manner, environmental indicators can be used to measure the environmental condition (health or quality) of a particular habitat, community or ecosystem (Hart *et al.* 1993; Hawksworth and Ritchie 1993). Lichens, for instance were used as indicators of air purity for at least a century because it was observed that in general, less lichens occurred in cities with polluted air than in non-polluted areas (de Wit 1983).

Environmental indicators include the types of indicators traditionally used in pollution studies, defined by Spellerberg 1991) and McGeoch (1998), for example as: (i) *Sentinels* - sensitive species introduced as "early warning devices" or to determine the effects of pollutants (the canaries used in early mining are probably one of the earliest uses of a "sentinel"); (ii) *Detectors* - species occurring naturally which show a measurable response to environmental change and can be usefully monitored for changes in behaviour, quantity or quality and (iii) *Exploiters* - species not normally present whose presence indicates (the probability of) disturbance or pollution. Hart *et al.* (1993) identified three types of indicators of environmental quality: *Detectors*, *Exploiters*, and taxon/taxa that are *normally present* and whose

absence reflects a change, as the third. Baseline data is essential to validate absences for all types of indicators.

Environmental indicators indicate a particular environmental condition or set of environmental conditions by change/no change in the indicator. Environmental indicators are used to describe, analyse and summarise environmental conditions, trends and their significance (Fisher 1998). Because of this focus on conditions and trends they are inexorably linked with monitoring programs (Fisher 1998) and state-of-the-environment reporting (Bertollo 1998; Saunders *et al.* 1998), since the prime objective in the application of environmental indicators is to assess environmental quality.

(2) Ecological indicators

However, as anthropogenic impacts on the environment increase, the need to protect and manage natural resources has become increasingly critical and the focus of environmental indication has expanded from stress-specific, end-of-pipe environmental quality standards to encompass the broad objectives of ecologically sustainable development: to protect biodiversity and to maintain ecological integrity (Cairns *et al.* 1993; Fisher 1998). To ensure that biological resources are managed on a sustainable basis requires increased emphasis on ecologically focused management and the monitoring of biological diversity and ecosystem functions. This has led to a perceived need for ecological indicators.

Ecological indicators have been defined as those environmental indicators that describe or represent the condition of an ecosystem or one of its critical components or processes (Bertollo 1998; Fisher 1998) including biological diversity and ecological processes (Vora 1997). However, McGeoch (1998) defines ecological indicators as environmental indicators whose response reflects that of at least a subset of other organisms; in other words, 'a characteristic or surrogate taxon/taxa for a community or ecosystem', or at least a subset of the larger community (Meffe and Carroll 1994).

The response of ecological indicators to environmental stressors is manifested by change/no change in the indicator. This response is used to measure the effects of anthropogenic stressors on the ecosystem and its components (Cairns *et al.* 1993; Bertollo 1998), in particular, its biotic communities (McGeoch 1998). Ecological indicators have particular applications notably in the areas of monitoring environmental changes and the effects of land management practices. To assess the

impacts of forest management in order to demonstrate that forest harvesting is not detrimental to the sustainability of wildlife populations and their habitats, for example (McLaren *et al.* 1998).

The critical difference between environmental indicators and ecological indicators is that environmental indication uses the response of biota to describe an environmental state or to detect anthropogenic perturbations, and, effects on the indicators themselves are of secondary importance; whereas ecological indication is the use of biota to determine the effect of an anthropogenic perturbation on the biota itself, and, the response of the indicators themselves is what is important. Baseline data or reference sites are important for both environmental and ecological indication, since change or no change can only be assessed when the characteristics of the biota are known, or where comparisons can be made with unimpacted biota in similar habitats elsewhere in the region.

(3) Biodiversity indicators

Biodiversity has become a major focus for conservation efforts (Noss 1990; McFadgen and Simpson 1994; Pearson 1994). Prioritisation and comparisons of key areas for conservation rely on estimates of biodiversity, yet this information is generally not readily available. Detailed knowledge of species distribution for most species is scarce, indeed, data on the absolute biodiversity for most areas are non-existent, particularly for invertebrates such as beetles, (Beccaloni and Gaston 1995; Balmford *et al.* 1996b). Moreover, the diversity of organisms and complexity of ecosystems prevent complete inventory even at the site scale (Kremen 1992; Hammond 1994).

Since all the species in a region can never be directly counted, practical evaluation of the relative biodiversity of an area or sets of areas must depend on surrogate information (Faith and Walker 1996a). If biodiversity is to be inventoried and monitored, measurable surrogate indicators of its composition, structure and functioning must be identified (Noss 1990; Kremen 1992; Pearson and Cassola 1992; Williams and Gaston 1994; Beccaloni and Gaston 1995; New 1995; Balmford *et al.* 1996a,b; McGeoch 1998).

Such surrogate taxa should be information rich; in other words, the pattern of species distributions in these taxa should correlate either with patterns of environmental heterogeneity or with distributional patterns of species in other taxonomic groups (Kremen 1994; Vane-Wright *et al.* 1994; Faith and Walker

1996b; Cranston and Trueman 1997). Biodiversity indicators therefore, are taxon or taxa whose diversity reflects some measure of the diversity (for example, character richness, species richness, level of endemism) of other taxa in a habitat or set of habitats (Noss 1990; Ryti 1992; Gaston and Blackburn 1995; Vane-Wright 1996; McGeoch 1998).

The primary application for biodiversity indicators is in reserve selection: to identify priority areas for conservation (Vane-Wright *et al.* 1991; McGeoch 1998) and to determine regional patterns of biodiversity (Pearson and Cassola 1992) in order to select a network of areas representative of the biodiversity of the region (Pressey *et al.* 1993; Margules *et al.* 1994; Faith and Walker 1996a). But they should also provide enhanced biogeographic resolution of communities, habitats and biotypes (Kremen *et al.* 1993). Biodiversity indicators save the time and expense that would be necessary for comprehensive surveys (if such surveys were at all possible) (McGeoch 1998).

ISSUES

'Useful indicators respond significantly to components one wishes to monitor'
(Elliott 1997 p 62).

The desirable qualities of indicators can differ depending on their purpose (Kremen *et al.* 1993; McGeoch 1998), and therefore, indicators cannot be selected until goals and objectives are specified (Cairns *et al.* 1993). The use of indicator taxa for conservation research has been divided in two categories: inventory studies and monitoring studies (Kremen *et al.* 1993; Pearson 1994; Rodríguez *et al.* 1998). Although inventory and monitoring are interrelated and both necessary for conservation, they frequently differ in their objectives and hence in the type of indicators to be used (Kremen *et al.* 1993). Inventory will often encompass the measurement of many things but monitoring should be specific to the planned objectives and assumptions used in planning (Vora 1997).

The main objectives of using indicators for inventory purposes are to provide information on regional patterns of biodiversity (Pearson and Cassola 1992) to identify priority areas for conservation (Vane-Wright *et al.* 1991; McGeoch 1998), to provide baseline data for the selection and planning of reserves; and to provide enhanced biogeographic resolution of communities, habitats and biotypes (Kremen *et al.* 1993). Therefore, indicators selected for inventory should be biodiversity indicators. Indicators for inventory should exhibit relatively high species diversity

and encompass the geographic range of interest (Kremen *et al.* 1993), but the most desirable characteristic of biodiversity indicators is that they accurately indicate/reflect aspects and patterns of diversity of other taxa/organisms or at least a subset at the relevant scale.

The objectives of monitoring programs are to evaluate changes in communities, habitats or ecosystems (structure, composition and function) resulting from environmental perturbations such as human disturbances, management activities or natural factors over time; to provide assessments of management and land use practices; to provide early warnings of ecological changes and to provide an accurate and sensitive source of information from which conditions and trends can be defined and recognised, and management decisions made (Noss 1990; Spellerberg 1991, 1992; Kremen 1992; Kremen *et al.* 1993).

Therefore, indicators selected for monitoring may be environmental, ecological or biodiversity indicators depending on the purpose of monitoring. Indicator assemblages appropriate for monitoring must be sensitive to environmental change, should exhibit a diversity of life-history and ecological preferences and should be able to provide an early warning of ecological change (Kremen *et al.* 1993). The most desirable characteristic of environmental indicators is that they accurately indicate a particular environmental condition. The most desirable characteristic of ecological and of biodiversity indicators is that they accurately indicate/reflect the response of other members of the community, habitat or ecosystem.

> **'Basically, everything is an indicator of something but nothing is an indicator of everything'** (Cairns *et al.* 1993 p. 6).

However, no single species or taxon can be expected to adequately represent or indicate patterns of diversity or function as a focal group for all other species and taxa (Hammond 1994; Pearson 1994). In addition, anthropogenic perturbations affect habitats across a range of scales, from microhabitat to landscape. Moreover, ecological investigations directed at conserving biological diversity span global, continental, regional and local scales (Humphries *et al.* 1995; Flather *et al.* 1997), in addition to organisms (plant and animal taxa) populations and communities at the local scale.

Therefore, we need to be able to identify biodiversity and to predict the potential impacts of human activities on biological diversity across a hierarchy of spatial and temporal scales (Hansen *et al.* 1991; White *et al.* 1997). And, since different

components of an ecosystem may respond differentially to stress, all major biological groups should be evaluated (Hart *et al.* 1993; Cottingham and Carpenter 1998). This means that a range of indicator taxa (plant, vertebrate, invertebrate), representing all major functional guilds and covering the full range of body sizes, in the system of interest, will be necessary to produce a reliable and consistent level of information across the range of scales (Di Castri *et al.* 1992; Kremen 1992; Kremen *et al.* 1993; McLaren *et al.* 1998; Whitford *et al.* 1998).

Hammond (1994) has termed this the "shopping basket" approach: the use of a representative selection of indicators that together serve as a composite focal group. In combination, the taxa in this 'shopping basket' may provide an adequate representation of the response of the community, habitat or ecosystem of interest to the stressor of interest (McGeoch 1998) and provide greater ecological understanding (New 1995).

Carabids as indicators

New (1995) suggests Arthropoda as one of 8 invertebrate groups which may be major foci for documenting invertebrate assemblages. As one of the more well characterized arthropod taxa, carabids have strong potential as indicators (Kremen *et al.* 1993; Lövei and Sunderland 1996; Rykken *et al.* 1997), and I will explore their potential in more detail in the following chapter. Kremen *et al.* (1993) suggest that documenting the distribution of selected arthropod taxa, such as carabids, will improve the spatial resolution of conservation planning for reserve selection and design, and a monitoring program that includes assemblages of terrestrial arthropods representing a diversity of taxa and/or functional groups will automatically broaden the scope of the environmental factors that can be perceived.

In addition, for management of natural areas, monitoring of terrestrial arthropod indicators, such as carabids, is more likely to provide early warnings of ecological changes (Pearson and Cassola 1992; Kremen *et al.* 1993; Oliver and Beattie 1993; Kremen 1994). Their short generation times will enable them to respond to environmental changes more rapidly than do vertebrate indicators, which may exhibit population responses that do not become evident until too late for proactive management (Kremen *et al.* 1993). Arthropod indicators may also be used to assay the effects of further fragmentation on natural areas that no longer support vertebrate indicator species (Kremen *et al.* 1993).

SELECTION OF INDICATORS

The literature regarding the desirable and necessary characteristics of indicators is extensive, numerous authors have discussed practical and biological criteria, and a number of characteristics are repeatedly suggested. In general, ideal indicator taxa should meet the following criteria (not in priority order). The authors suggesting specific criterion are indicated by the letters following, and author details are given at the end of all listed criterion.

1. ***Play an important role in the ecosystem*** (a, b, d, f, g, i, l, m, o, p)

- Be functionally important at all trophic levels in ecosystem or habitat such that they integrate a wide variety of biotic and abiotic variables.
- Interact in many ways with other parts of the ecosystem, so that they are reliable indicators of the general environment in which they occur.
- occupy high trophic level or at least reflect some aspect of ecosystem function

2. ***Be diverse and abundant*** (a, b, c, d, f, l, p)

- with reasonable numbers species at any location, so that changes due to environmental conditions can be noted.
- although the number of species at a locality should be large but not so large as to be unmanageable (c; d).
- represent a range of body sizes (Stork 1990).

3. ***Be responsive to habitat variables and sensitive to environmental change/perturbations on a convenient and detectable scale*** (a, b, c, d, f, g, i, l, n, o)

- show a prompt and accurate response to a particular cause of perturbation
- and give insights into and mechanisms underlying such change so as to provide interpretable results

4. ***Be widely distributed*** (a, b, c, d, e, f, g, h, i, k, l, m, n, o, p)

- occur over a broad geographical range and breadth of habitat types so that results will be broadly applicable
- to allow for comparisons of trends or data from the same species in different locations

- but include local endemics, critical faunas or 'hotspots of diversity in selected areas (New 1994).

5. ***Be characteristic of the ecosystem/ habitat it is desired to assess or monitor*** (a, b, c, d, e, f, g, h, i, k, l, m, n, o, p)

- be expected in the area to be studied from knowledge of their biogeography and ecology.
- include a high proportion of specialist species, since specialisation of each population within a narrow habitat is likely to make them sensitive to habitat change
- a species presence at an important locality even if it is not distributed elsewhere may be an important source of information about that locality (g)

6. ***Be easily collected and sampled*** (a, b, c, d, e, f, g, h, k, l, m, n, o)

- so practical to use
- amenable to the application of portable and standardised sampling methods operated by non-specialists

7. ***Be readily identifiable*** (a, b, c, d, f, g, i, l, o, p)

- at species level so therefore practical to use.
- especially by people who are not experts in its taxonomy (g)
- ideally keys and field guides usable by non-specialist should be available or capable of being produced easily (c)
- readily identifiable to the level necessary for bioindication in the field with minimal equipment (i)

8. ***Taxonomically well known and stable*** (e, c, h, k, l, m, n, p)

- with an established taxonomic framework for all, or significant sections

- with most species described or recognisable, so that populations can be reliably defined

9. ***Biology and general life history well understood*** (e, h, k)

- with limiting resources, enemies, physical tolerances, and all stages of the life cycle available to readily incorporate into hypotheses and experimental design

10. ***Have potential economic importance to agriculture, environment etc.*** (e, h, k, m, p)

- so that scientists and politicians, especially in developing countries where pure or basic science is frequently considered a luxury, can be convinced that this taxon is worth dedicating local personnel and resources for studies
- 'values' including range of 'commodity' or 'applied' aspects likely to engender political support

11. ***Some evidence that patterns observed in the indicator taxon are reflected in other related and unrelated taxa*** (e, h, k)

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a, Majer 1983; b, Greenslade and Greenslade 1984; c, Hellawell 1986; d, Cranston 1990; e, Noss 1990; f, Andersen 1990; g, Lower and Kendall 1990; h, Pearson and Cassola 1992; i, Hawksworth and Ritchie 1993; j, Johnson *et al.* 1993; k, Pearson 1994; l, New 1993; m, New 1995; n, Saunders *et al.* 1998; o, Cairns *et al.* 1993; p, Stork 1994.

The next chapter focuses on these characteristics in the context of Carabidae.

SUMMARY

Conservation is in crisis mode as biodiversity declines at a rapid rate and anthropogenic impacts escalate. There are two options: inventory (a quick and dirty survey at the very least) and monitor all biota or use indicator taxa - a subset of the biota -to inventory and monitor. In Australia, due to resource limitations, and megadiversity, option one, to inventory and monitor all biota is just not possible. This leaves us with option two. However, it is also clear that no single species or taxon can be expected to adequately represent or indicate patterns of diversity or function as a focal group for all other species and taxa.

To adequately identify biodiversity and to predict the potential impacts of human activities on biological diversity across a hierarchy of spatial and temporal scales, a range of indicator taxa will be necessary. In combination, the taxa in this 'shopping basket' may then provide an adequate representation of the response of the community, habitat or ecosystem of interest to the stressor of interest and provide greater ecological understanding.

Beetles are a major part of invertebrate and total biodiversity. Strategies that fail to include invertebrates such as beetles, are likely to fail to conserve this, a major portion of all biodiversity. Not only should beetles and other invertebrates be considered in conservation strategies, but it is likely that carabid beetles, and other selected arthropods, may provide easy, cost effective, and sensitive means to measure and monitor the overall health of biodiversity, the state of the environment and the effects of anthropogenic stresses on biodiversity and ecosystems (Cairns *et al.* 1993; Noss 1990; Kremen *et al.* 1993; New 1995; McGeoch 1998). The taxa in our 'shopping basket' must contain representatives of invertebrate groups such as beetles. I will explore the potential of carabid beetles as a representative for other beetles in the following chapters.

Chapter Three

Assessing the indicator potential of Carabids: Carabids as environmental indicators

While the characteristics of a good indicator will be largely dependent on the context in which they are to be used and the objectives of the environmental issue being addressed, it is important that the potential of any proposed indicator taxon be demonstrated (Pearson and Cassola 1992; Pearson 1994; McGeoch 1998). This chapter assesses the suitability of ground beetles as indicator organisms by comparing the characteristics and attributes of carabid beetles against the template of attributes proposed in the previous chapter.

TEMPLATE OF ATTRIBUTES

1. ***Play an important role in the ecosystem.***

Carabids occupy an important trophic role in many terrestrial ecosystems (Thiele 1977; Stork 1990; Lövei and Sunderland 1996. As predatory organisms, the energy turnover of carabids in the ecosystem equals that of two other important animal groups of the soil surface, ie. chilopods and spiders (Thiele 1977). Carabids are among the most important polyphagous invertebrate predators of the soil fauna particularly in temperate regions (Loreau 1990).

2. ***Be diverse and abundant***

With over 40,000 described species world-wide, the Carabidae comprise more species than any other animal family (Thiele 1977; Evans and Bellamy 1996). Carabid assemblages are moderately species rich, usually with no more than 10-40 species active in a habitat in the same season, although regional assemblages are correspondingly richer (Britton 1991; Lövei and Sunderland 1996).

The prime requirement for a good indicator is that there are 'reasonable' numbers of species at any location and that these numbers are sufficient to indicate changes in environmental conditions (Majer 1983; Greenslade and Greenslade 1984; New

1995). Although carabid species richness in individual habitat types may be modest in Australia compared for example with Europe (New 1998), small numbers may still be sufficient to determine changes in environmental conditions. For example, an assemblage of only six species of lizard was considered adequate to detect differences in regeneration of sand-mined areas in a study by Twigg and Fox (1991). While Melbourne *et al.* (1997) demonstrated that changes in the abundance of four species of crickets and in five species of slugs were sufficient to demonstrate effect of grassland type in a survey of grassland types in the Australian Capital Territory. Further, small numbers (but sufficient to detect differences) may be an advantage, since large numbers may be unmanageable and more costly to assess (Hellawell 1986; Cranston 1990).

3. ***Be responsive to habitat variables and sensitive to environmental change/perturbations on a convenient and detectable scale***

Carabids can and have been used as environmental indicators for assessments of environmental pollution (Frietag 1979; Heliovaara and Vaisanen 1993) and habitat classification for nature conservation (Luff *et al.* 1992; Desender *et al.* 1991a; Pizzolotto 1994; Desender 1996) (see also criterion 5).

Carabids have been used in two distinct contexts in relation to responses to pollution (Frietag 1979): the effects of industrial pollutants on the environment, and the application of pesticides - mainly to phytophagous insects of which carabids are predators. Frietag assessed the future role of carabids as pollution indicators as "boundless", given their biological features and the wide range of chemical parameters to which they variously respond.

Comparative studies on community composition and variation have demonstrated that carabids respond to environmental and habitat variables, including management, at a range of scales. At the local scale, variations in carabid community composition has been shown to be significantly correlated with soil water content (Thiele 1977; den Boer 1986; Epstein and Kulman 1990; Holmes *et al.* 1993b; Luff *et al.* 1989), soil properties (Thiele 1977; Michaels and Mendel 1998), altitude (Thiele 1977; Butterfield and Coulson 1983; Luff *et al.* 1989; Holmes *et al.* 1993b; Michaels and Mendel 1998) and land management practices (Thiele 1977; Eyre and Rushton 1989; Holmes *et al.* 1993a; Niemelä *et al.* 1993a; Michaels and McQuillan 1995).

Carabid beetle communities have been shown to reflect differences in grassland management, by responding to changes in the substrate resulting from agricultural practices (Eyre *et al.* 1989) . Other studies have also shown that grassland carabids respond to management regimes, with some species favouring intensively managed sites and others being very sensitive to any changes to their normal natural regimes (Eyre and Rushton 1989; Rushton *et al.* 1989;Eyre *et al.* 1990). Asteraki *et al.* (1995) found variations in the number of carabid species in grassland sites to be dependant on the type of grassland field margin with many preferring hedges to fences. While Rushton *et al.* 1990) found grazing regimes to be an important factor influencing carabid assemblages in grasslands.

At a regional scale, carabid assemblages respond to various factors including management, isolation and habitat fragmentation. Intensively managed sites were shown to form distinct groups within classifications of grasslands based on carabid species data at both regional and national scales (Eyre *et al.* 1986b). Based on changes in carabid assemblages from 69 locations representing a range (9) of natural and sub-natural open habitat types in Belgium, Dufrene and Legendre (1997) noted that isolation and habitat fragmentation influenced species distribution. They concluded that their results indicated the potential of carabids for local-scale assessment and biological assessment.

At a larger geographical scale, spatial variations in the distribution of 61 local faunas of the genus *Carabus* L. across the Russian Plain, have been linked to two major groups of environmental factors: regional, influenced by historical reasons such as glaciations and isolations; and zonal, resulting from recent climatic conditions (Penev 1996). Other studies have also demonstrated that carabid distributions respond to climatic variations. For example, Ashworth (1996) stated that there was ample fossil evidence to demonstrate that shifts in geographic ranges of carabids occurred as a result of climate change.

4. ***Be widely distributed***

Carabids are represented on a more or less global basis (Erwin *et al.* 1979; Britton 1991; Lövei and Sunderland 1996). From forests to grasslands, mountaintops to seashores, structural, physiological and behavioural adaptations have enabled carabids to invade all types of habitats on all continents, with the exceptions of deserts, (where carabids are limited to streams and oasis, (Erwin *et al.* 1979)) and antarctic regions. In the studies used as the basis for this thesis, carabids were present in all sites sampled.

5. ***Be characteristic of the ecosystem/ habitat it is desired to assess or monitor***

Classification of habitats based on carabid data have shown that characteristic assemblages of carabids can be related to particular ecosystems and habitats. An analysis of data sets comprising many year-cycle pitfall trap catches of carabids in different habitats in Belgium (including forests, hedges, fenlands, moors, grasslands and coastal and aquatic and semi aquatic vegetation), revealed clear differences in species distributions between habitat classes (Dufrene *et al.* 1990). Ordination based on similarities between sampling sites demonstrated that the habitat preferences of carabid communities and distinctions between habitat classes could be identified by the individual species distribution. Similarly, in an investigation of five different dune habitats in the State Nature Reserve, De Westhoek on the Belgium coast, Maelfait and Desender (1990) found the carabid faunas of the different habitat types clearly identifiable, and suggested that carabid beetles could be used in site assessment of dune ecosystems to evaluate and monitor temporal changes in these habitats and to assess their nature conservation value.

Classification and ordination of ground beetle data in north-east England by Luff *et al.* (1989) showed that ten ecologically meaningful habitat groups: coastal (1), upland (2), woodland (1), grassland (2), riverside (3) and marsh (1) could be recognised by their carabid communities. While Eyre *et al.* 1996) used ground beetle survey data to produce distinct habitat classifications for riverside habitats using disparate data sets (namely a compilation of riverside ground beetles species lists from throughout the UK and Ireland; records from pitfall trapping in Scotland and local lists from a survey of one riverside in England.

Other quantitative investigations on the distribution of carabid beetles in various habitats have established species preferences for particular habitat types and demonstrated that it is possible to recognise communities of carabids typical of certain habitats such as forests (Thiele 1977; den Boer 1986; Buse 1988; Niemelä *et al.* 1992; Michaels and Mendel 1998), peatlands (Bauer 1989; Holmes *et al.* 1993a,b), mires (Jonsell 1995), grasslands (Butterfield and Coulson 1983; Eyre and Luff 1990), wet heath (Michaels and Mendel 1998), and limestone grasslands and juncus moor and blanket peat (Bauer 1989). In addition, many carabid species have adapted to specialised habitats (Thiele 1977). For example, some *Nebria* spp. live at the edge of glaciers (Erwin 1985) and ecological specialisation in Tasmania and Australia includes a substantial troglobiont fauna (Eberhard *et al.* 1991; New 1998). These results suggest that ground beetles have considerable potential in

environmental and conservation assessments as determinants of environmental state and quality.

6. *Be easily collected and sampled*

Carabids are easily and cheaply sampled with modest effort and without the use of specialised equipment or personnel by using pitfall traps (see Chapter Four). Moreover, carabids are also amenable to mark and recapture experiments. For example by Wallin (1986) to examine micro habitat preferences and by Mader *et al.* (1990) in a study of linear barriers to arthropod movements in the landscape. Radio telemetry is also being used to study dispersal and habitat use (Riecken and Raths 1996), although at present the use of small radio transmitters to track movement is restricted to very large carabid species which are able to carry these relatively heavy instruments.

7. *Be readily identifiable*

Carabids have an easy-to recognise generalist body plan, and the genera are generally very distinctive and, with experience, can be recognised at a glance. At the species level, carabids can generally be separated using simple morphological features. Sexual dimorphism is not usually extreme (Sloane 1920). Even with difficult species it is rarely necessary to resort to inconspicous characters such as the reproductive apparatus, but when it is, the male genitalia offer excellent diagnostic characteristics. While there is still a considerable amount of work to be done on the immature stages, matching juveniles with adults is not really a problem, since there are generally sufficient adults trapped to make it unnecessary and captures of larvae are less common since they are predominantly soil dwellers. While the taxonomy of many insects at the species level is extremely poorly known, carabids are one of the best studied groups and even where species cannot be confidently named, they can generally, be readily assigned to genera.

8. *Be taxonomically well known and stable*

Carabid taxonomy has been extensively studied over the last 200 years and the basic classification is well established (Moore *et al.* 1987). In the temperate zone of the Northern Hemisphere, there have been extensive integrated studies of carabid biology and ecology beginning with Lindroth over the period 1945-1949 and extended by Thiele (1977) and Erwin *et al.* (1979).

The modern understanding of the Australian carabid fauna was developed by T. G. Sloane in a series of over 50 papers published over the period 1888-1933, and by P.J. Darlington Jr. in a series of papers from 1943 to 1961, and extended by B. P. Moore in over 30 papers since 1960. Although not the largest beetle family in Australia, the Australian carabid fauna listed in the Zoological Catalogue of Australia includes about 1,800 described species with at least some representatives of most of the major global tribes (Moore *et al.* 1987). Recent revisionary studies principally by M. Baehr and B.P. Moore have extended the described fauna to around 2000 species (New 1998) and provided notable advances to the knowledge of several carabid groups (including some of the smaller and more cryptic groups such as the Bembidiinae, Lebiinae and Trechini), their distribution, and keys to species level identification. Although, as noted by Moore *et al.* (1987), many of the larger genera are yet to be the subject of revisionary studies and hence many species may still have additional synonyms.

9. *Biology and general life history well understood*

Carabids are one of the most studied invertebrate families in the world. The biology and general life history of many species are known and the ecology of some species is known in detail. The corpus of published work on the Carabidae and especially their role in communities is impressive. For example, the results of the First International Symposium of Carabidology held in 1976 were published by Erwin *et al.* (1979), subsequent proceedings were published by den Boer *et al.* (1986), and den Boer *et al.* (1987); and the role of carabids in environmental and ecological studies was explored by numerous authors in Stork (1990). A quick check on Current Contents for Life Sciences revealed over 500 references for the last eight years. A quarterly journal, *Carabologia*, and a carabid website (Ground Beetles http://henry.ento.cornell.edu/CUIC/carabid_lst.htm) serve as a forum for both amateur and professional carabidologists.

Pearson (1994) suggests that the breadth of studies available on a potential indicator taxon around the world, including review articles, newsletters and journals dedicated to the biology of the group, would serve as evidence of well known biology and natural history. There would appear then to be ample evidence that carabids fit this criterion. Admittedly, compared to the breadth of work on carabids elsewhere, particularly in Europe, the Australian carabid fauna is considerably less studied and therefore readily available biological and ecological information is consequently meagre at present. However, there is a growing body of work on carabids in Australia: for example, on the effects of logging (Michaels and

McQuillan 1995); in the monitoring and assessment of biodiversity (Oliver and Beattie 1996a,b; Davies and Margules 1998; New 1998; Michaels 1999); the fidelity of plant and carabid assemblages (Oliver *et al.* 1998; Michaels and Mendel 1998); biological control (Calver *et al.* 1986; Robertson 1986); and studies on ecophysiological characteristics of individual species such as those by Horne 1990; Horne 1992a,b on two species of *Notonomus* in Victoria.

10. ***Have potential economic importance***

While the potential economic importance of carabids have yet to be truly exploited, there is no doubt that carabids fit this criterion. Predators such as carabids play an important role in regulating populations of many phytophagous pests (Hance 1987; Clark *et al.* 1997). Both adult and larval carabids have been found to be potentially important natural pest control agents in agroecosystems (Sunderland and Vickerman 1980; Clark *et al.* 1994; Symondson 1994; Clark *et al.* 1997). For example, Sunderland and Vickerman (1980) showed they were important predators of aphids in cereals. Clark *et al.* (1997) found four species to be potential predators of a variety of agronomic pests. *Pterostichus melanarius* (Illiger), for example, predates on aphids in sugar beet and cereal systems (Clark *et al.* 1997). Symondson (1994) demonstrated that *Abax parallelipipedus* (Piller and Mitterpacher) was highly effective at reducing the numbers of slugs and therefore, crop damage, and hence could be used as a biological control agent for slugs.

Experiments in the field and in the laboratory have also revealed that carabids possess a regulatory effect on harmful insects. For example, the introduction of *Calosoma sycophanta* into North America had a regulatory effect on the destructive moths that had been brought in from the Old World (Thiele 1977). In south-western Australia, carabids were found to be predators of the bush fly, *Musca vetustissima* (Walker) (Diptera: Muscidae) (Calver *et al.* 1986). Experimental studies of predation on grassland populations of Australian soldier fly, *Inopus rubriceps* (Macquart) (Diptera: Stratiomyidae) demonstrated that predation by carabids influenced the size of the subsequent larval population (Robertson 1986).

11. ***Some evidence that patterns observed in the indicator taxon are reflected in other related or unrelated taxa***

To date, there is little available evidence that carabids fit this criterion, although it has been suggested that as predators in the soil invertebrate food chain, carabids can

be used as indicators of other invertebrate species (Allen 1979; Hengeveld 1980; Rykken *et al.* 1997).

CONCLUSION

It has been demonstrated that, as indicators, carabids fit all but the last *a priori* criteria. However, testing against these criteria merely minimises the possibility of selecting an inappropriate taxon (McGeoch 1998). It is obviously vital in all categories that an accepted sampling protocol is available and that they are readily identifiable. Cost effectiveness will also be a major consideration, since one of the prime reasons given for needing indicators is lack of resources. However, the applicability of other criteria will obviously depend on the category of indicator and some criteria may be scenario-specific or optional. For example, information on life history may not be necessary for biodiversity assessments (Kremen 1992) and wide range may be optional (McGeoch 1998). In fact, a wide range is unlikely to be useful if the taxon is not present in the habitat of interest. A case in point is tiger beetles (Cicindelidae), a widespread taxon on a global basis (Pearson and Cassola 1992), but absent in Tasmania. Although suggested as a potential biodiversity and ecological indicator (Pearson and Cassola 1992; Rodríguez *et al.* 1998), its absence would preclude its use in either role in Tasmania.

Once a taxon is identified as potentially suitable, it should be tested for indicator value (Pearson 1994; McGeoch 1998). Many taxa have been suggested as potential indicators, but there has been little formal testing of taxa identified as suitable based on the above *a priori* criteria. It has been suggested that the basis for selection is often merely favoured or convenient taxa (Soulé and Kohm 1989; Woiwod and Thomas 1993; Williams and Gaston 1994; Pearson 1994). Indeed, vegetation has long been used as an indicator because it is convenient and well resourced. In particular, although some authors have proposed fulfilment of some of the above criteria for indicator species as the reason as to why their own specialist group is a good indicator, either of diversity or environmental change, the last criterion, "some evidence that patterns observed in the indicator taxon are reflected in other related or unrelated taxa" has rarely been addressed.

Carabids are a very appropriate taxon to test the validity of this component of the indicator concept. It is demonstrated that they fit all but one of the *a priori* criteria (not just some) and they are a subset (a large and well known one) of a very substantial subset of the (global) biota. But, if they are to qualify as an ecological indicator or a biodiversity indicator (*sensu* McGeoch 1998), their representativeness

of other taxa must be demonstrated. The extent to which carabids meet this last criterion will be addressed in further chapters.

Chapter Four

Study sites

The data used in this thesis was obtained from several different studies undertaken at a range of locations in Tasmania. In total, there are 73 sites in four biomes: dry sclerophyll forests, wet sclerophyll forests, coastal sand dunes and grasslands (Fig.4.1).

Sites in the dry sclerophyll forest in the Eastern Tiers and Fingal Tiers, remnant native grassland sites and coastal sand dune sites were used to evaluate the potential of carabids as biodiversity indicators for other selected Coleoptera.

Sites in the wet sclerophyll forest in the Picton Valley and sites in dry sclerophyll forest in Weilangta State Forest were used to evaluate the potential of carabids as ecological indicators for other selected Coleoptera.

This chapter describes the location, vegetation, and other environmental parameters for each of the study sites.

SITE VARIABLES

Site codes

Each site was given a short site code to facilitate processing, archiving and analysis. Site codes incorporate various information such as location, plant community type, dominant plant species, or the regeneration age and are used when individual sites are mentioned in this thesis.

Vegetation sampling

A vegetation survey was conducted at each site and the vascular plant species recorded. All species within 5 metres of the pitfall trap transect lines were recorded. A general reconnaissance was also undertaken to identify species not recorded along transect lines. Vegetation surveys at sites in the Eastern and Fingal Tiers and the Grasslands were conducted by botanists: Dr Louise Mendel and Ms Louise Gilfedder respectively. At all other sites the vegetation was identified *in situ* using personal experience, keys and flora manuals. Difficult specimens were collected,

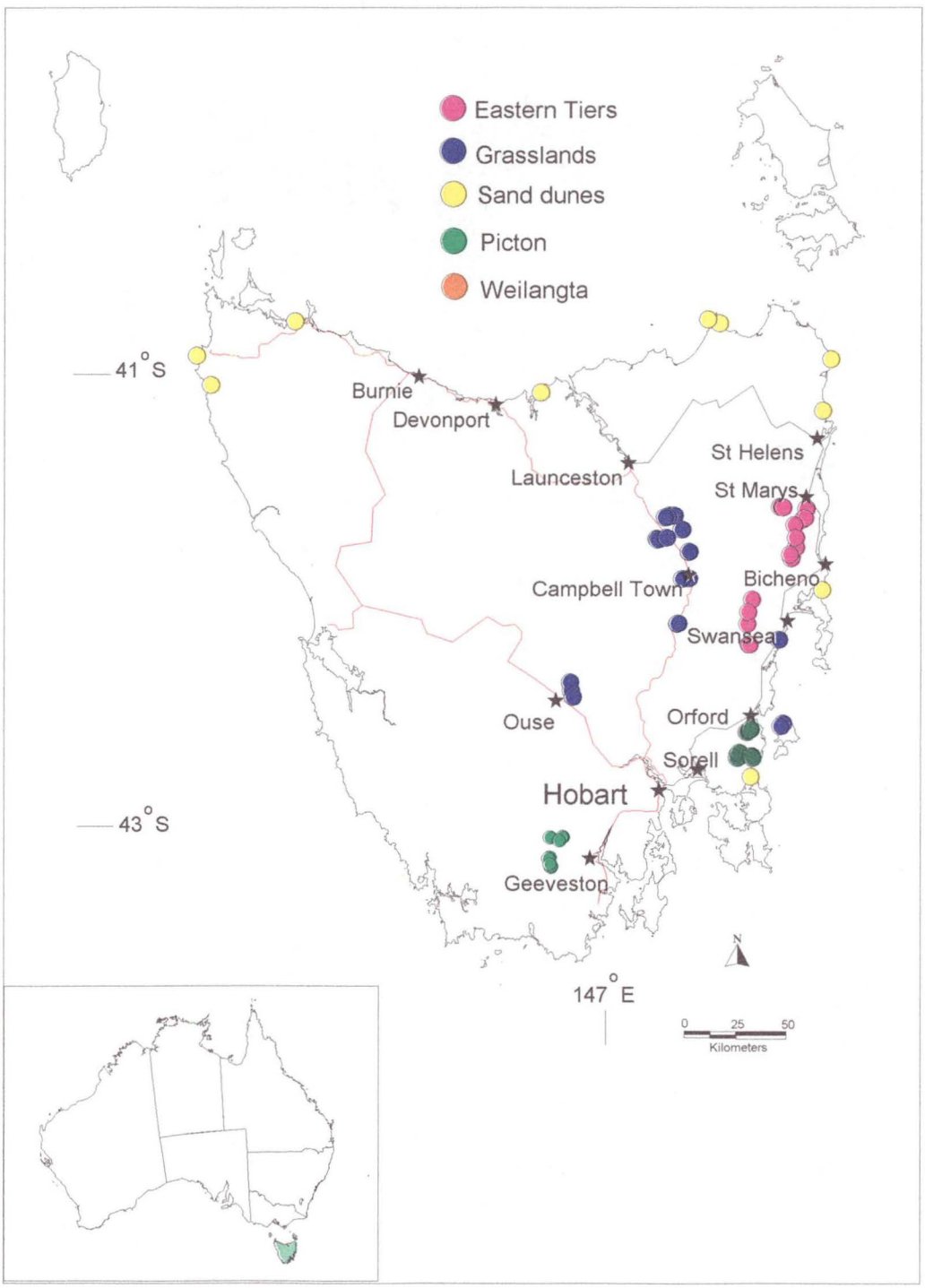


Figure 4.1. Geographic locations of study areas in Tasmania. ● = sampling sites.

pressed and either subsequently identified to species by botanists or given a morphospecies name.

Environmental variables

A range of physical variables were measured at each site. The first 6 were recorded for all sites, other variables were recorded for selected groups of sites.

- Slope (using a clinometer) was measured in degrees from the horizontal.
- Aspect (using a compass). Aspect was then coded from 1 to 5, from the driest north-west facing slopes to the wettest south-east facing slopes (after Kirkpatrick and Nunez 1980). 1 = NW, 2 =N or W, 3 = NE or SW, 4 = E or S, and 5 = SE.
- Elevation (using 1:25000 maps) recorded to the nearest 10m above AHD (Australian Height Datum).
- Soil acidity (pH): recorded to the nearest 0.5 unit, measured using a CSIRO pH kit.
- Geology (by referencing the appropriate Land Systems of Tasmania).
- Soil texture: determined by analysing the relative proportions of sand silt and clay within the soil. For ANOVA, % of clay and % sand were used. Soil cores to a depth of 11 cm were used to measure texture, stoniness, pH and organic matter. Six cores were taken at each site and mean values were used in the analysis.
- Soil stoniness: determined by calculating the percentage volume of stone in the soil samples and ordered from 1 to 5 for ANOVA: 1= rare:<1%, 2 =slightly stony: 1-5%, 3 = stony: 5-20%, 4 = very stony: 20-50%, 5 = extremely stony: 50-75% (Marsden and Allison 1992). Soil texture and stoniness provide an indication of the drainage properties and permeability of the substrate.
- Organic matter content was determined by loss on ignition (LOI).
- Tussockness: was a subjective measure: 0 = no tussocks, 1 = sparse low tussocks, 2 = dense low tussocks, 3 = sparse tall tussocks, 4 = dense tall tussocks
- Grazing: ordered from 1 to 5 for ANOVA (1 = ungrazed, UG, 2 = marsupial, Ma, 3 = slight, S, 4 = moderate, M, and 5 = heavy, H). Sites were grazed by various combinations of sheep (the main grazers on private land), deer, cattle and native marsupials.

SITE DESCRIPTIONS

EASTERN TIERS

- *The Eastern Tiers and Fingal Tiers: dry sclerophyll forest (and wet heath)*

- *18 sites*

This group of sites was located in the predominantly dry sclerophyll forests of the Eastern Tiers and Fingal Tiers on the east coast of Tasmania (Fig. 4.2.). The Eastern Tiers range between Fingal and Tooms Lake and the Fingal Tiers between Avoca and St Marys and west of the Douglas Apsley National Park. The annual rainfall ranges from 750 to 1000 mm distributed rather evenly over the year, although evaporation can be strong in summer The geology is predominantly Jurassic dolerite which supports relatively fertile soils rich in clay, but of variable permeability (Davies 1988).

Six different plant communities were selected on the basis of representativeness, and differences in dominant species composition by a botanist (Dr Louise Mendel). Three replicate sites were established within each of the six community types. While there are variations specific to site, the plant communities broadly fall within those described by Duncan and Brown 1985), and Kirkpatrick *et al.* 1988).

Site location details and plant community type are given in Table 4.1. Table 4.2 lists the six botanical communities (PCTs) chosen, and summarises the typical vegetation found within each plant community type. Other environmental variables are shown in Table 4.3. Examples of sites within each plant community type are shown in Figures 4.6 to 4.8.

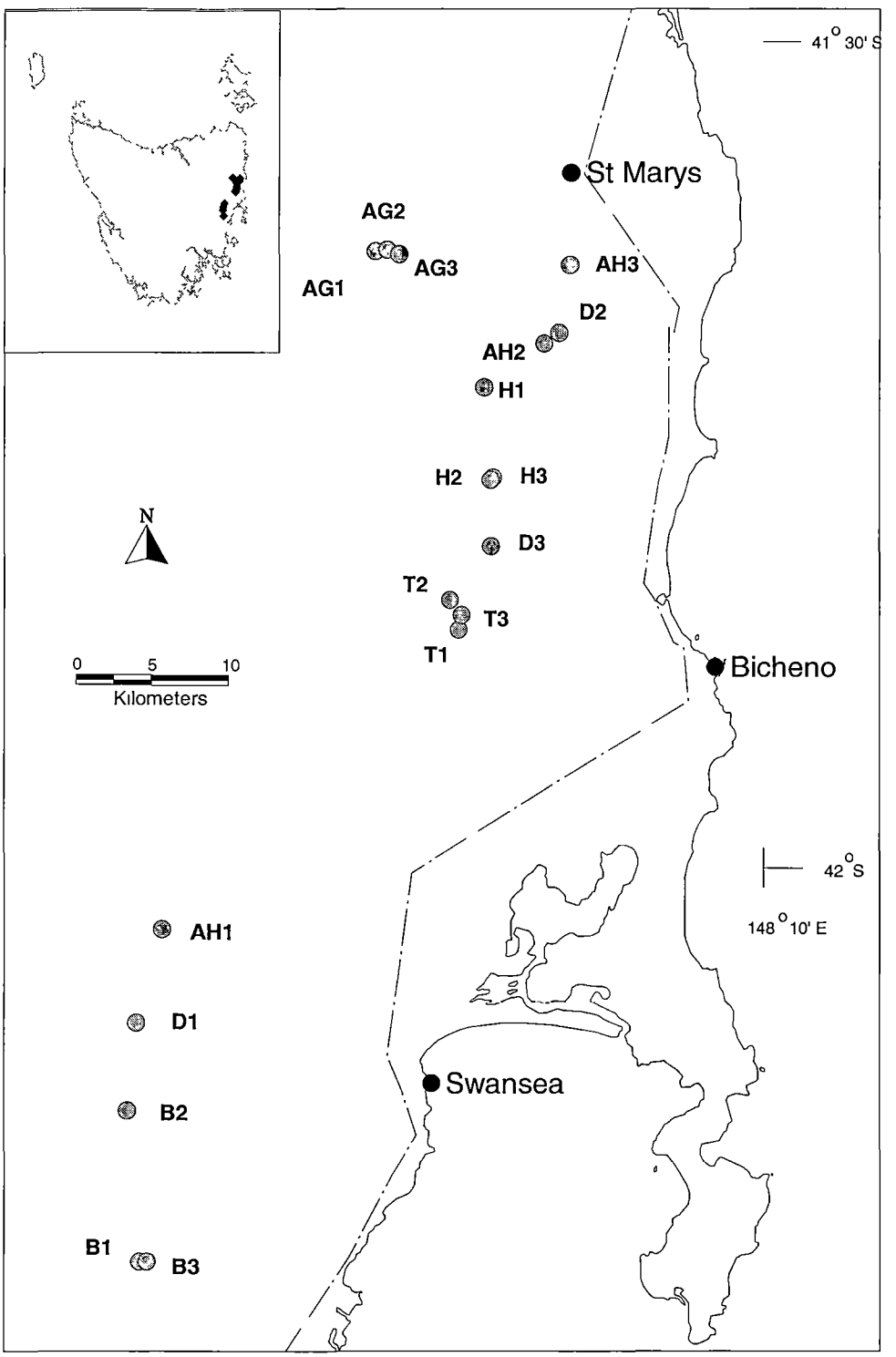


Figure 4.2. Geographical locations of sites in the dry sclerophyll forests of the Eastern and Fingal Tiers. Letters indicate the plant community type.

Site No.	Code	Locality	Plant Community Tpe	Map reference	Easting	Northing
1	AG1	Mt Fosters	*E. amygdalina* grassy open forest	St Marys 5839	585500	5390700
2	AG2	Mt Fosters	*E. amygdalina* grassy open forest	St Marys 5839	586300	5390800
3	AG3	Mt Fosters	*E. amygdalina* grassy open forest	St Marys 5839	587100	5390500
4	AH1	Tooms Lake	*E. amygdalina* heathy open forest	Leake 5634	571500	5345500
5	AH2	Douglas Apsley	*E. amygdalina* heathy open forest	Fingal 5838	596800	5384600
6	AH3	Douglas Apsley	*E. amygdalina* heathy open forest	Fingal 5838	598500	5389800
7	B1	Tooms Lake	*E. brookerana* wet sclerophyll forest	Tooms 5632	570000	5323400
8	B2	Tooms Lake	*E. brookerana* wet sclerophyll forest	Tooms 5632	569200	5333500
9	B3	Tooms Lake	*E. brookerana* wet sclerophyll forest	Tooms 5632	570500	5323400
10	D1	Tooms Lake	*E. delegatensis* shrubby open forest	Colonels 5633	569800	5339300
11	D2	Douglas Apsley	*E. delegatensis* shrubby open forest	Fingal 5839	597800	5385300
12	D3	Harding Falls	*E. delegatensis* shrubby open forest	St John 5837	593300	5371100
13	H1	Mt Fosters	Wet heath	Fingal 5838	592800	5381700
14	H2	Coal Marsh	Wet heath	St John 5837	593400	5375700
15	H3	Coal Marsh	Wet heath	St John 5837	593400	5375800
16	T1	Harding Falls	*E. tenuiramis* heathy open forest	Henry 5836	591100	5365500
17	T2	Harding Falls	*E. tenuiramis* heathy open forest	Henry 5836	590500	5367500
18	T3	Harding Falls	*E. tenuiramis* heathy open forest	Henry 5836	591300	5366500

Table 4.1. Site location details for sites in the Eastern Tiers: site numbers, site codes, locality, plant community type and location data (TASMAP 1:25000 reference).

PCT	Canopy	shrub and low tree layer	low shrub layer	sedges and grasses	ground layer
AG	*Eucalyptus amygdalina* grassy open forest				
	Dominant *Eucalyptus amygdalina* Sub-dominant *Eucalyptus viminalis* (25-30m)	*Acacia dealbata* *Exocarpus cupressiformis* *Pomaderris apetala* *Bursaria spinosa* (8-12m)	*Epacris impressa* *Comosperma volubile* *Coprosma quadrifida*	*Lomandra longifolia* *Lepidosperma* spp. *Poa labillardieri* *Poa rodwayi* *Themeda triandra* *Stipa* spp.	*Acaena novae-zelandiae* *Viola hederacea* *Goodenia lanata* *Oxalis perennans* *Wahlenbergia* spp. *Geranium solanderi*
AH	*Eucalyptus amygdalina* heathy open forest				
	Dominant *E. amygdalina* Sub-dominant *E viminalis* *Eucalyptus delegatensis* (25-35m)	*A dealbata* *Banksia marginata* *B. spinosa* (6-15m)	*Lomatia tinctoria* *Cyathodes glauca* *Cyathodes parvifolia* *Pultenaea* spp	*Gahnia grandis* *L. longifolia* *Lepidosperma* spp.	*A. novae-zelandiae* *V hederacea* *Hibbertia riparia* *Pimelea humilis*
B	*Eucalyptus brookerana* wet sclerophyll forest				
	Dominant *Eucalyptus brookerana* Sub-dominant *Eucalyptus pulchella* *E. delegatensis* (20-25m)	*A dealbata* *P. apetala* *Bedfordia salicina* *Ziera arborescens* *Leptospermum scoparium* (7-12m)	*C. quadrifida* *Clematis aristata*	*G. grandis* *Dicksonia antarctica*	*A. novae-zelandiae* *V. hederacea* *Hypochoeris radicata* *Oxalis* spp.
D	*Eucalyptus delegatensis* shrubby open forest				
	Dominant *E delegatensis* Sub-dominant *E. viminalis* *E. amygdalina* (25-35m)	*A. dealbata* *B marginata* *B. salicina* (8-12m)	*L. tinctoria* *C. glauca* *Hakea lissosperma* *Tetratheca pilosa*	*G. grandis* *L. longifolia* *Lepidosperma* spp	*G. solanderi* *V. hederacea*
H	Wet Heath				
		Hakea epiglottis *Leptospermum lanigerum* *Sprengelia incarnata* (1 5-2.5m)	*Epacris gunnii* *Epacris lanuginosa* *Baurea rubiodes*	*G. grandis* *Lepidosperma filiforme*	*Centella cordifolia* *Empodisma minus* *Restio australis*
T	*Eucalyptus tenuiramis* heathy open forest				
	Dominant *E tenuiramis* Sub-dominant *E. viminalis* *E delegatensis* (15-25m)	*A. dealbata* *B. marginata* *Allocasuarina littoralis* (8-10m)	*L scoparium* *Amperea xiphoclada* *Pultenaea gunnii* *L. tinctoria*	*G. grandis* *Pteridium esculentum* *L longifolia* *P. rodwayi*	*Goodenia lanata* *P. humilis* *Gnaphalium collinum*

Table 4.2. The six representative botanical communities chosen in the Eastern Tiers study: typical vegetation, floristics and heights, at sites within each plant community type (PCT).

Site	S (°)	Asp.	Elv.	pH	ST	SS	OC (%)
AG1	10	NE	300	5.5	L	4	17
AG2	12	NNW	300	5.0	SL	4	22
AG3	14	NNE	300	6.0	L	4	14
AH1	2	W	600	5.5	CL	3	21
AH2	10	NNW	450	6.0	SICL	2	25
AH3	11	NNW	350	6.0	SL	4	13
B1	14	WNW	550	5.5	SL	4	11
B2	25	ESE	550	6.0	SL	3	32
B3	10	SSE	550	6.0	SL	1	11
D1	19	SSE	350	6.5	SL	2	12
D2	9	N	500	5.5	SL	2	16
D3	12	ESE	400	6.0	SL	5	21
H1	3	S	500	5.5	L	4	17
H2	5	S	500	5.5	LS	1	46
H3	4	N	500	4.5	SL	1	17
T1	5	E	200	6.0	SL	5	18
T2	18	ENE	200	6.0	L	5	15
T3	1	NE	200	6.0	SL	5	13

Table 4.3. Environmental parameters for sites in the Eastern Tiers. S = slope; Asp. = aspect; Elv. = elevation (m.a.s.l.); pH = pH of surface horizon; ST = soil texture (CL = clay loam, L = loam, LS = loamy sand, SL = sandy loam, SICL = silty clay loam); SS = soil stoniness (1= rare:<1%, 2 =slightly stony: 1-5%, 3 = stony: 5-20%, 4 = very stony: 20-50%, 5 = extremely stony; 50-75%); OC = organic content (100 - %loss on ignition) of surface horizon.

Fig. 4.3. *Eucalyptus amygdalina* grassy open forest in the Eastern Tiers, site AG3.

Fig. 4.4. *Eucalyptus amygdalina* heathy open forest in the Eastern Tiers, site AH1.

Fig. 4.5. *Eucalyptus brookerana* wet sclerophyll forest in the Eastern Tiers, site B2.

Fig. 4.6. *Eucalyptus delegatensis* shrubby open forest in the Eastern Tiers, site D2.

Fig. 4.7. Wet Heath in the Eastern Tiers, site H3.

Fig. 4.8. *Eucalyptus tenuiramis* heathy open forest in the Eastern Tiers, site T2.

GRASSLANDS

- *Remnant native grasslands located throughout the Midlands, on the east coast and on Maria Island.*

- *23 sites*

This group consisted of twenty three remnant native grassland sites located throughout the Midlands, on the east coast of Tasmania and on Maria Island (Fig. 4.9).

Sites were chosen in consultation with a botanist (Ms Louise Gilfedder) as good representative examples of six of the eight grassland community types (GCT's) classified and described by Kirkpatrick *et al.* 1995) and McDougall and Kirkpatrick 1994).

Site location details and grassland community type are given in Table 4.4. Table 4.5 lists the six grassland community types and summarises the typical vegetation species within each grassland community type. Other environmental variables are shown in Table 4.6. Examples of sites within the grassland community types are shown in Figures 4.10 to 4.15.

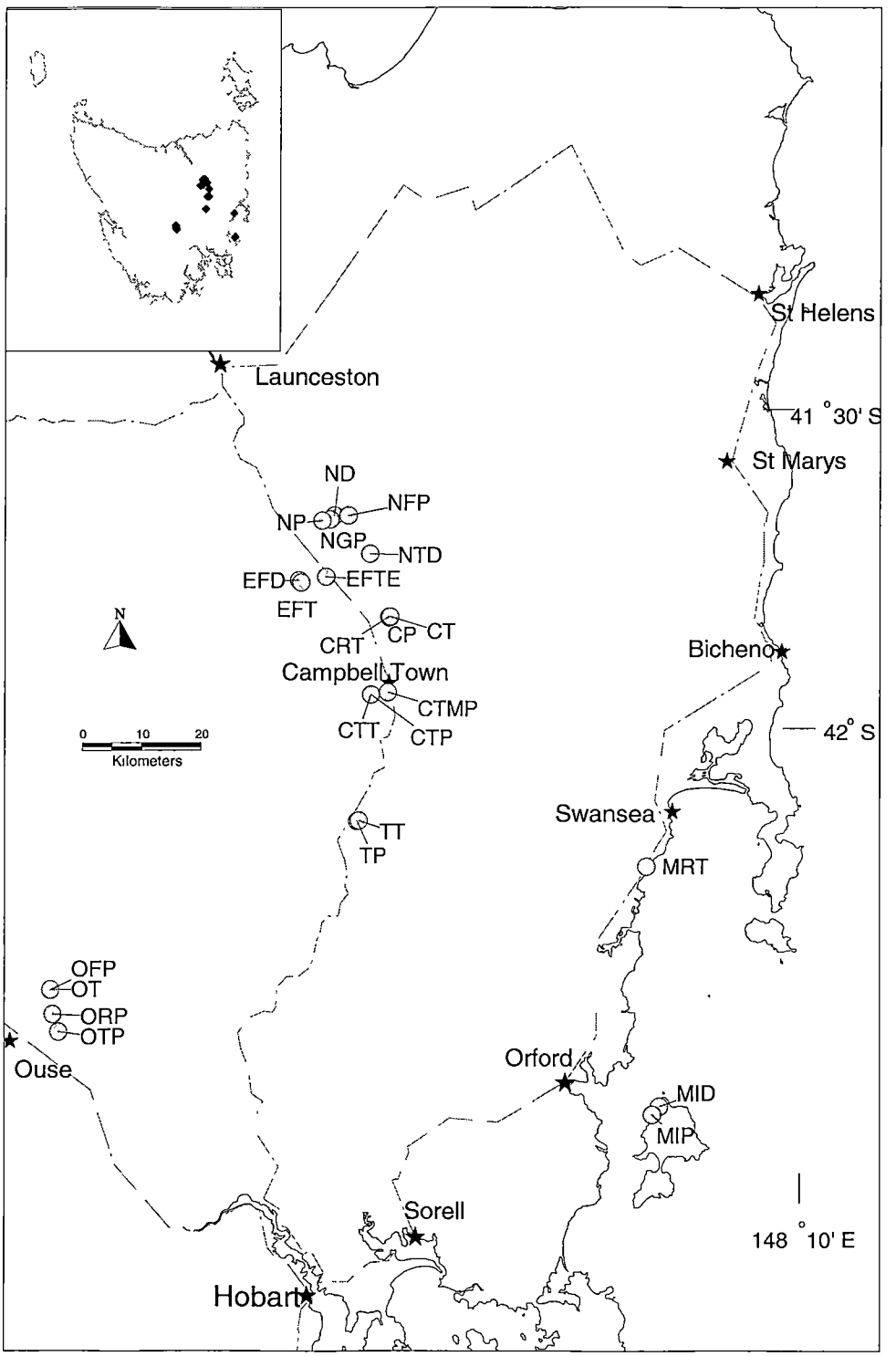


Figure 4.9. Geographical locations of remnant native grassland sites in the Grasslands study Letters indicate the locality and the dominant vegetation type of the grassland community for each site.

Site No.	Code	Locality	GCT	Map reference	Easting	Northing
1	ORP	Ouse	T1	Cawood	483300	5300500
2	OTP	Ouse	T1	Montacute	484300	5297500
3	CP	Conara	T2	Diamond	540800	5368700
4	CTMP	Campbelltown	T2	Campbelltown	540500	5355800
5	CTP	Campbelltown	T2	Jacobs	537600	5355400
6	MIP	Maria Island	T2	Darlington	585900	5283300
7	NGP	Nile	T2	Nile	530500	5385700
8	TP	Tunbridge	T2	Tunbridge	535200	5333500
9	NFP	Nile	T3	Nile	533600	5386400
10	NP	Nile	T3	Nile	529100	5385500
11	OFP	Ouse	T3	Cawood	482900	5305600
12	CRT	Conara	T5	Diamond	540700	536900
13	CT	Conara	T5	Diamond	540900	5368900
14	CTT	Campbelltown	T5	Jacobs	537600	5355400
15	EFT	Epping Forest	T6	Cleveland	525500	5374800
16	EFTE	Epping Forest	T6	Cleveland	529800	5375800
17	MRT	Mayfield Bay	T6	Mayfield	584900	5325700
18	OT	Ouse	T6	Cawood	482900	5304700
19	TT	Tunbridge	T6	Tunbridge	535500	5333600
20	EFD	Epping Forest	T8	Cleveland	525100	5375200
21	MID	Maria Island	T8	Darlington	587100	5284700
22	ND	Nile	T8	Nile	531200	5386400
23	NTD	Nile	T8	Cleveland	537400	5379800

Table 4.4. Site location details for Grassland sites: site numbers, site codes, locality, grassland community type (GCT) and location data (TASMAP 1:25000 reference).

GCT	GCT description	Characteristic species	Common species
T1	Tussock Grass-Sagg-Buzzies Tasmanian Valley Grassland		
	Dense stands dominated by tall Poa and Sagg tussocks	*Poa labillardierei* (Tussock Grass) *Lomandra longifolia* (Sagg) *Danthonia* sp. (Wallaby Grass) *Acaena novae-zelandiae* (Buzzies)	*Ehrharta stipoides* *Cymbonotus preisssianus* *Agrostis capillaris*[i] *Hypochoeris radicata*[i]
T2	Tussock Grass-Weeping Grass-Flat Daisy Tasmanian Valley Grassland		
	Sparse Poa tussocks interspersed with a sward of grasses and herbs	*P. labillardierei* *Danthonia* spp. *Themeda triandra* (Kangaroo Grass) *Solenogyne dominii* (Flat Daisy)	*Stipa* spp. *Asperula conferta* *E. stipoides* *H. radicata*[i]
T3	Tussock Grass-Rush-Matted St John's Wort Tasmanian Flood Plain Grassland		
	Large Poa tussocks interspersed by rank grasses graminoids and herbs	*P. labillardierei* *J. australis* (rushes) *Carex breviculmus* (sedge) *Oxalis perennans* (Oxalis)	*A. echinata* *Dichondrra repens* *Plantago varia* *H. radicata*[i]
T5	Kangaroo Grass-Hirsute Guinea-flower-Urn Heath Tasmanian Grassland		
	Dense low Themeda tussocks with a rich intertussock flora of herbs, shrubs and other grasses	*Themeda triandra* *Hibbertia riparia* Guinea-flower) *Lissanthe strigosa* (Urn Heath) *Bossiaea prostrata* (Prostrate Bossiaea)	*Schoenus absconditis* *Hypericum gramineum* *O. perennans* *E. stipoides*
T6	Kangaroo Grass-Tasmanian-Spear Grass-Common Everlasting Tasmanian Grassland		
	Open Themeda tussocks grassland with a rich intertussock flora of grasses and herbs	*Themeda triandra* *Poa rodwayi* (Tussock Grass) *Stipa stuposa* (Tasmanian Spear Grass) *Chrysocephalum apiculatum* *Convulvulus erubescens*	*Geranium solanderi* *H. gramineum* *O. perennans* *Danthonia* spp. *H. radicata*[i]
T8	Wallaby Grass-Native Cranberry East Coast Tasmanian Grassland		
	Sparse tussocks dominated by Wallaby Grass with a range of other grasses, herbs and prostrate shrubs	*Danthonia* spp. *P. rodwayi* *Wahlenbergia gracilis* (Bluebells) *Dichondrra repens* *O. perennans*	*T. triandra* *Eryngium vesiculosum* *Gnaphalium collinum* *Solenogyne dominii* *E. stipoides*

Table 4.5. Vegetation characteristics at Grassland sites Grassland community type (GCT), description, and typical vegetation at sites within each GCT. Characteristic species are species used to define the GCT; Common species occurred at all sites within the GCT. i = introduced species.

Code	S(°)	Asp.	Elv.	pH	Geo	ST	SS	OC%	GR	T
ORP	0	N	250	6	JD	SL	3	14	UG	4
OTP	6	N	330	7 5	JD	SCL	4	29	S	4
CP	0	N	210	5.5.	QS	SL	2	15	H	1
CTMP	0	N	200	6	TB	LS	3	17	S	1
CTP	0	N	180	5.5	TB	LS	2	18	M	2
MIP	4	NW	20	6	JD	SCL	3	18	Ma	3
NGP	0	N	170	5.5	QS	SCL	2	10	M	3
TP	12	E	200	9.5	TS	C	4	20	UG	4
NFP	2	SSW	220	6	JD	C	3	16	M	3
NP	0	N	160	6 5	QS	SCL	2	15	S	4
OFP	0	N	160	6	TS	LS	2	9	UG	4
CRT	0	N	210	6.5	QS	SL	4	23	UG	2
CT	0	N	210	6 5	QS	LS	2	19	H	0
CTT	0	N	180	6	TB	SCL	2	16	M	1
EFT	3	WNW	220	6	QS	SL	2	11	M	1
EFTE	0	N	160	5.5	QS	LS	4	16	UG	2
MRT	2	W	20	7	JD	SCL	3	8	UG	2
OT	0	N	160	6	TS	LS	4	16	H	0
TT	0	N	200	6	TS	SCL	3	20	UG	2
EFD	5	WNW	200	5	QS	SCL	1	16	M	0
MID	5	SW	20	6	JD	SCL	4	16	Ma	1
ND	2	N	180	6	QS	SL	2	6	H	0
NTD	5	S	215	5.5	QS	SCL	1	16	M	1

Table 4.6. Environmental parameters for sites in the Grasslands. S = slope; Asp. = aspect; Elv. = elevation (m.a.s.l.); Geo = Geology (JD = Jurassic dolerite, QS = Quaternary sediments, TB = Tertiary basalt, TS = Triassic sandstone) (Davies 1988); ST = soil texture (LS = loamy sand, SL = sandy loam; SCL = sandy clay loam, C = clay); SS = soil stoniness (1= rare:<1%, 2 =slightly stony: 1-5%, 3 = stony: 5-20%, 4 = very stony: 20-50%, 5 = extremely stony; 50-75%); pH = pH of surface horizon; OC = organic content (100 - %loss on ignition) of surface horizon; GR = grazing regime: (UG = ungrazed, Ma = marsupial, S = slight, M = moderate, and H = heavy); T = tussockness: (0 = no tussocks, 1 = sparse low tussocks, 2 = dense low tussocks, 3 = sparse tall tussocks, 4 = dense tall tussocks).

Fig. 4.10. Grassland community type T2, Tussock Grass-Weeping Grass-Flat Daisy Tasmanian Valley Grassland, site MIP, Maria Island.

Fig. 4.11. Grassland community type T2, Tussock Grass-Weeping Grass-Flat Daisy Tasmanian Valley Grassland, site TP, Tunbridge.

Fig. 4.12. Grassland community type T3, Tussock Grass-Rush-Matted St John's Wort Tasmanian Flood Plain Grassland, site NFP, Nile.

Fig. 4.13. Grassland community type T5, Kangaroo Grass-Hirsute Guinea-flower-Urn Heath Tasmanian Grassland, site CT, Conara.

Fig. 4.14. Grassland community type T6, Kangaroo Grass-Tasmanian –Spear Grass-Common Everlasting Tasmanian Grassland, site EFT, Epping Forest.

Fig. 4.15. Grassland community type T8, Wallaby Grass-Native Cranberry East Coast Tasmanian Grassland, site EFD, Epping Forest.

SANDDUNES

- *West, north and east coasts of Tasmania: coastal sand dunes*

- *10 sites*

Ten sites representative of medium to high energy sandy beaches were sampled in this group of sites. Two sites on the north west coast, four on the north coast and four on the east coast of Tasmania (Fig. 4.16).

Rainfall for the eastern and southern beaches ranges from 700-800mm fairly evenly spread throughout the year. Rainfall for the northern and western beaches ranges from 750-1500mm with a winter maximum.

Table 4.7 summarises the typical vegetation species occurring at each site. Site location details are given in Table 4.8. Environmental variables are shown in Table 4.9. A selection of sites including one from each coast, are shown in Figures 4.17 to 4.20.

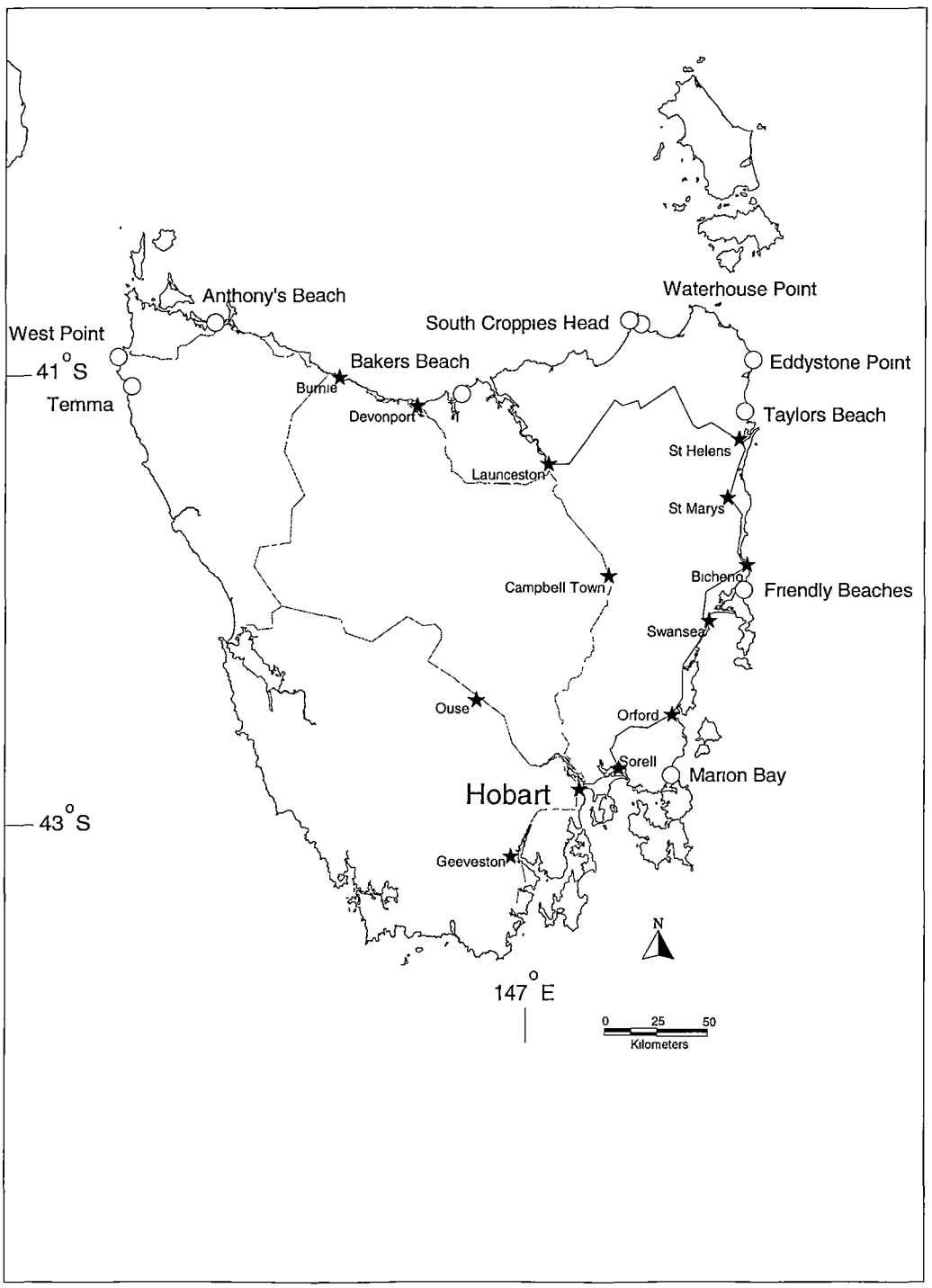


Figure 4 16. Geographic location of the ten coastal sand dune sites.

Site	Small tree layer	shrub layer	Grasses and sedges	ground layer
AB	*Acacia sophorae* *Banksia marginata*	*Leucopogon parviflora*	*Lomandra longifolia* *Pteridium esculentum* *Ammophila arenaria*[1] *Juncus pallidus*	*Acaena novae-zelandiae*
FB	*Acacia sophorae* *Banksia marginata* *Allocasuarina monolifera*	*Leucopogon parviflora* *Leptospermum laevigatum*	*Austrofestuca littoralis* *Lomandra longifolia* *Pteridium esculentum*	*Carpobrotus rossii* *Lupinus arboreus*[1]
WH	*Acacia sophorae* *Allocasuarina monolifera*	*Leucopogon parviflora* *Cyathodes abietina* *Pimela flava*	*Lomandra longifolia* *Spinifex hirsutus* *Pteridium esculentum*	*Lupinus arboreus*[1]
MB	*Acacia sophorae* *Banksia marginata*	*Lupinus* spp.[1]	*Austrofestuca littoralis* *Lepidosperma concavum* *Lomandra longifolia*	*Carpobrotus rossii*
SC	*Acacia sophorae* *Banksia marginata*	*Leucopogon parviflora* *Helichrysum* *Pimelea flava* *Coprosma repens*[1]	*Austrofestuca littoralis* *Lomandra longifolia* *Spinifex hirsutus* *Pterisium esculentum*	*Carpobrotus rossii* *Alyxia buxifolia (seabox)*
WP	*Acacia sophorae*	*Leucopogon parviflora*	*Ammophila arenaria*[1] *Juncus pallidus*	*Carpobrotus rossii* *Acaena novae-zelandiae*
TB	*Acacia sophorae* *Acacia axillaris* *Banksia marginata* *Allocasuarina monolifera*	*Leucopogon parviflora* *Dilewynia glaberrima*	*Austrofestuca littoralis* *Lomandra longifolia*	*Carpobrotus rossii*
EP	*Acacia sophorae*	*Leucopogon parviflora* *Helichrysum*	*Austrofestuca littoralis* *Lomandra longifolia* *Spinifex hirsutus* *Pteridium esculentum*	*Carpobrotus rossii* *Acaena novae-zelandiae*
BB	*Acacia sophorae*	*Leucopogon parviflora* *Helichrysum* *Pimelea flava*	*Lomandra longifolia* *Spinifex hirsutus* *Pteridium esculentum*	*Lupinus arboreus*[1]
TE			*Lepidosperma concavum* *Juncus pallidus*	*Acaena novae-zelandiae*

Table 4.7. Typical vegetation floristics at each of the ten sites in the Sanddunes study. i = introduced.

Site No.	Code	Locality	Coast	Map Reference	Easting	Northing
1	AB	Anthony's Beach	North coast	Stanley 3448	346800	5481200
2	FB	Friendly Beaches	East coast	Friendly 6034	606100	5346200
3	WH	Waterhouse Point	North coast	Waterhouse 5447	555600	5480200
4	MB	Marion Bay	East coast	Dunalley 5625	571100	5258500
5	SC	South Croppies Head	North coast	Waterhouse 5447	550200	5476200
6	WP	West Point	West coast	Marrawah 3046	299700	5464400
7	TB	Taylors Beach	East coast	Binalong 6043	606700	5437600
8	EP	Eddystone Point	East coast	Eddystone 6046	601600	5463500
9	BB	Bakers Beach	North coast	Port Sorell 4644	468200	5446200
10	TE	Temma	West coast	Temma 3043	306200	5437300

Table 4.8. Site location details for sites in the Sanddunes: site numbers, site codes, locality, coastal location and location data (TASMAP 1:25000 reference).

Site	S (°): Beach	S (°): Dune	Asp.	pH	GR
MB	21	36	NE	7.5	Ma
FB	18	29	ENE	7.5	Ma
TB	8	21	NE	7	Ma
EP	28	38	E	7.5	Ma
WH	12	14	N	8	Ma
SC	21	27	NW	8	Ma
BB	33	59	NW	8	Ma
AB	9	13	N	8	H
WP	11	23	S	8.5	Ma
TE	24	28	NW	8.5	H

Table 4.9. Environmental parameters for sites in the Sanddunes. S = slope; Asp = aspect; pH = pH of surface horizon; GR = grazing regime: (UG = ungrazed, , Ma = marsupial, , S = slight, M = moderate, and H= heavy).

Fig. 4.17. Sanddunes: Friendly Beaches, site FB.

Fig. 4.18. Sanddunes: Waterhouse Point, site WH.

Fig. 4.19. Sanddunes: Bakers Beach, site BB.

Fig. 4.20. Sanddunes: West Point, site WP.

PICTON

- *The Picton Valley: wet sclerophyll forest*

- *10 sites*

Sites in this group are located in the Picton Valley, 100km south of Hobart and directly west of the Hartz mountains (Fig. 4.21). It is a broad low elevation (100-300m a.s.l.) valley of rather uniform, tall mixed forest largely dominated by *Eucalyptus obliqua.* Cool temperate forest, dominated by *Nothofagus cunninghamii*, with emergent *E. obliqua*, is present in some gullies.

Geology in the area ranges from Triassic-Permian sandstone, formed on sediments of the Parmeneer supergroup to Jurassic dolerite (Davies 1988). Annual rainfall totals are approximately 1100 mm distributed evenly throughout the year. Much of the area has been subjected to selective logging over the last century and to the current forestry practice of clearfell and slash-burn since the 1970's.

Sites consisted of a chronosequence of logged coupes; 1, 3, 9 and 25 years duplicated on the basis of regeneration age, along with two old-growth forest sites, a total of 10 sites.

Site location and forest class details are given in Table 4.10. Table 4.11 details the environmental variables and Table 4.12 summarises the typical vegetation within each age class. Representative examples of sites within each of the forest classes are shown in Figures 4.22 to 4.25.

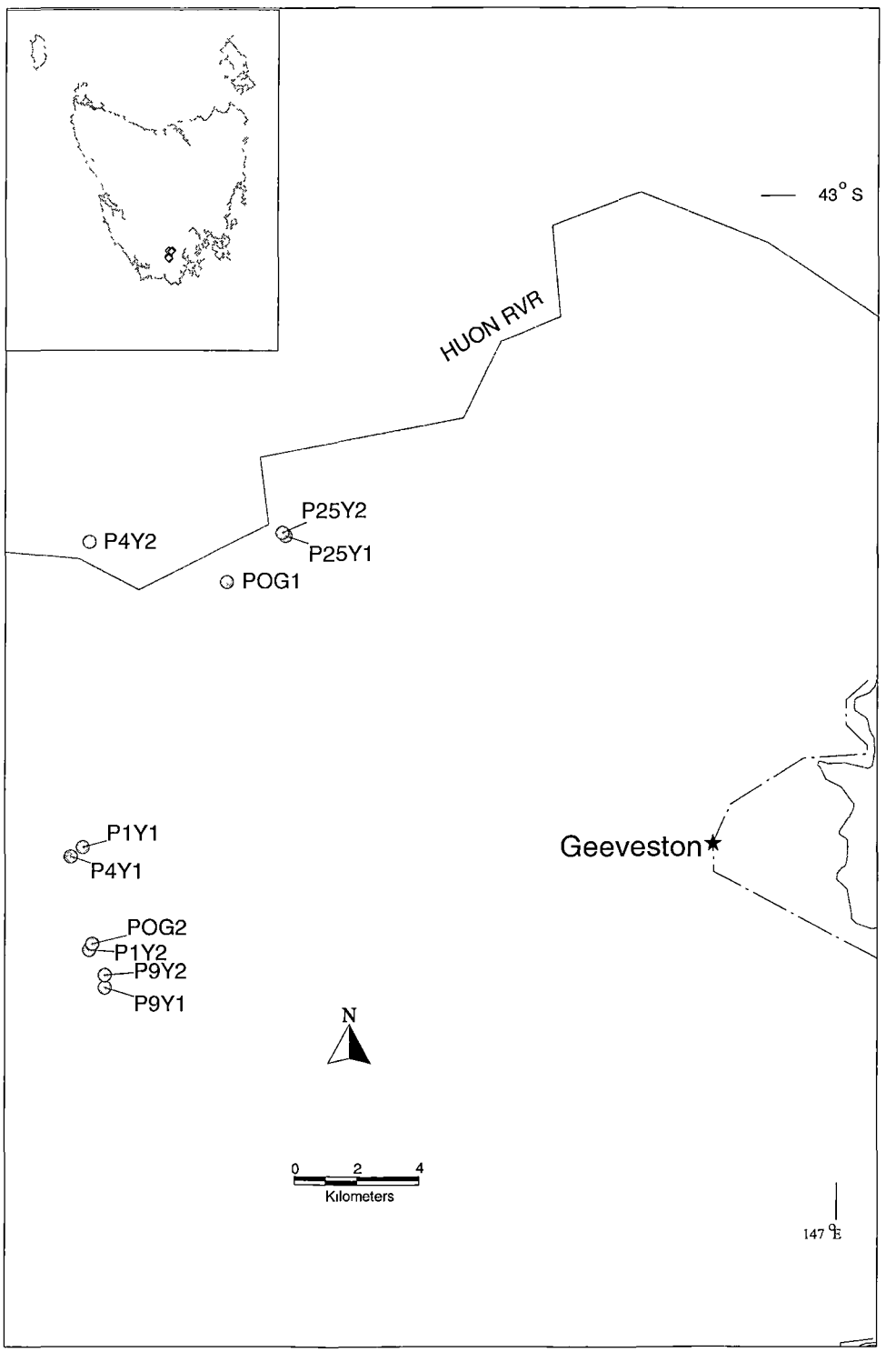


Figure 4.21. Geographical location of study sites in the Picton.

Code	Locality	Age	Regen. class	Map reference	Easting	Northing
P1Y1	Picton Spur	1	Early	Burgess	473000	5218800
P1Y2	Farmhouse Creek	1	Early	Burgess	473200	5215500
P4Y1	Picton Spur	4	Early	Burgess	472600	5218500
P4Y2	Tahune	4	Early	Picton	473200	5228700
P9Y1	Farmhouse Creek	9	Intermediate	Burgess	473700	5214300
P9Y2	Farmhouse Creek	9	Intermediate	Burgess	473700	5214700
P25Y1	Tahune	25	Late	Picton	479500	5228900
P25Y2	Tahune	25	Late	Picton	479400	5229000
POG1	Tahune	OG	Old-growth	Picton	477600	5227400
POG2	Farmhouse Creek	OG	Old-growth	Burgess	473300	5215700

Table 4.10. Site location details for the Picton: site codes, locality, age (years since clearfell; OG = old-growth control), Regen. class = regeneration classification as used in text of this thesis, and location data (TASMAP 1:25000 reference).

Site	S(°)	Asp.	Elv.	Geo	ST	OC%	No traps
P1Y1	9	ESE	200	JD	SL	0.04	10
P1Y2	14	N	300	JD	SL	0.05	6
P3Y1	14	ESE	200	JD	SL	0.07	6
P3Y2	8	S	200	JD	LS	0.06	10
P9Y1	18	E	300	JD	SCL	0.09	6
P9Y2	22	E	300	JD	SCL	0.12	10
P20Y1	8	NW	100	TPS	SCL	0.08	10
P20Y2	8	WSW	100	TPS	SC	0.08	6
POG1	7	NNW	100	TPS	SCL	0.05	6
POG2	22	ENE	300	JD	LS	0.02	10

Table 4.11. Environmental parameters for sites in the Picton. S = slope; Asp. = aspect; Elv. = elevation (m.a.s.l.); pH = pH of surface horizon; Geo = Geology (JD = Jurassic dolerite, TPS = Triassic-Permian sandstone); ST = soil texture (LS = loamy sand, SL = sandy loam; SCL = sandy clay loam, SC = sandy clay); OC = organic content (100 - %loss on ignition) of surface horizon. No. of traps = the trapping effort used at each site (see Chapter Five).

Forest class	Vegetation description
Early (1 yr)	Little vascular vegetation, except for a few widely scattered clumps of *Gahnia grandis*. There is a considerable amount of logs and charred log debris.
Early (4 yr)	Vegetation consists of saplings of *Eucalyptus obliqua*, 2-3m tall, interspersed with *Pomaderris apetala*, 1-2 m in height, with a ground cover of liverworts and mosses. Also scattered seedlings of *Anopterus glandulosus* and *Acacia dealbata*, 1-2m high, and dense clumps of *Gahnia grandis*.
Intermediate (9 yr)	The vegetation is dominated by *Eucalyptus obliqua* to 10m in height. *Ziera arborescens*, *Cyathodes juniperina*, *Nothofagus cunninghamii*, *Phyllocladus asplenifolius* and *Eucryphia lucida*, from 1-5m in height form the midstorey, and dense clumps of *Gahnia grandis* dominate the understorey.
Late (25 yr)	*Eucalyptus obliqua* to 30m in height, with a dense closed canopy, dominates the vegetation. The midstorey consists of *Nothofagus cunninghamii*, *Cyathodes glauca* and *Pommaderris apetala*, to 15 m in height, with an understorey of declining *Gahnia grandis*, and a groundcover of dense leaf litter and mosses.
Old-Growth	Vegetation is characterized by scattered *Eucalyptus obliqua* to 60m in height,with a midstorey of *Nothofagus cunninghamii*, *Anopterus glandulosus*, and *Eucryphia lucida*; and an understorey of *Gahnia grandis*, with a groundcover of mosses.

Table 4.12. Typical vegetation at sites within each forest age class in the Picton.

Fig. 4.22. Early regrowth site in the Picton, site P4yr1.

Fig. 4.23. Intermediate regrowth site in the Picton, site P9yr1.

Fig. 4.24. Late regrowth site in the Picton, site P25yr1.

Fig. 4.25. Old-growth control site in the Picton, site OG1.

WEILANGTA

- *Weilangta State Forest: dry sclerophyll forest*

- *12 sites*

This group of sites was located in the Weilangta State Forest, approximately 60 km north of Hobart on the east coast of Tasmania, directly south of Orford and east of Nugent (Fig. 4.26) The Weilangta State Forest contains a wide range of vegetation types including *Eucalyptus obliqua* tall forest, *Eucalyptus delegatensis* tall forest and coastal grassy forest, but can be broadly described as dry sclerophyll forest, with an elevational range of 200- 650m.

Most of the area consists of rugged Jurassic dolerite hills with a small area of hilly Triassic sandstone immediately south of Prossers Sugarloaf (Davies 1988). The annual rainfall ranges from 625mm to 750mm. Much of the area has been subjected to selective logging over the last century and to the current forestry practice of clearfell and slash-burn since the 1970's.

Sites consisted of a chronosequence of logged coupes; 1, 9 and 20 years, each duplicated on the basis of regeneration age class, along with six old-growth sites, a total of 12 sites.

Site location and forest class details are given in Table 4.13. Environmental variables are shown in Table 4.14, and Table 4.15 summarises the typical vegetation at sites within each forest age class. Examples of Early regrowth, Intermediate regrowth and Old-growth sites are shown in Figures 4.27 to 4.29.

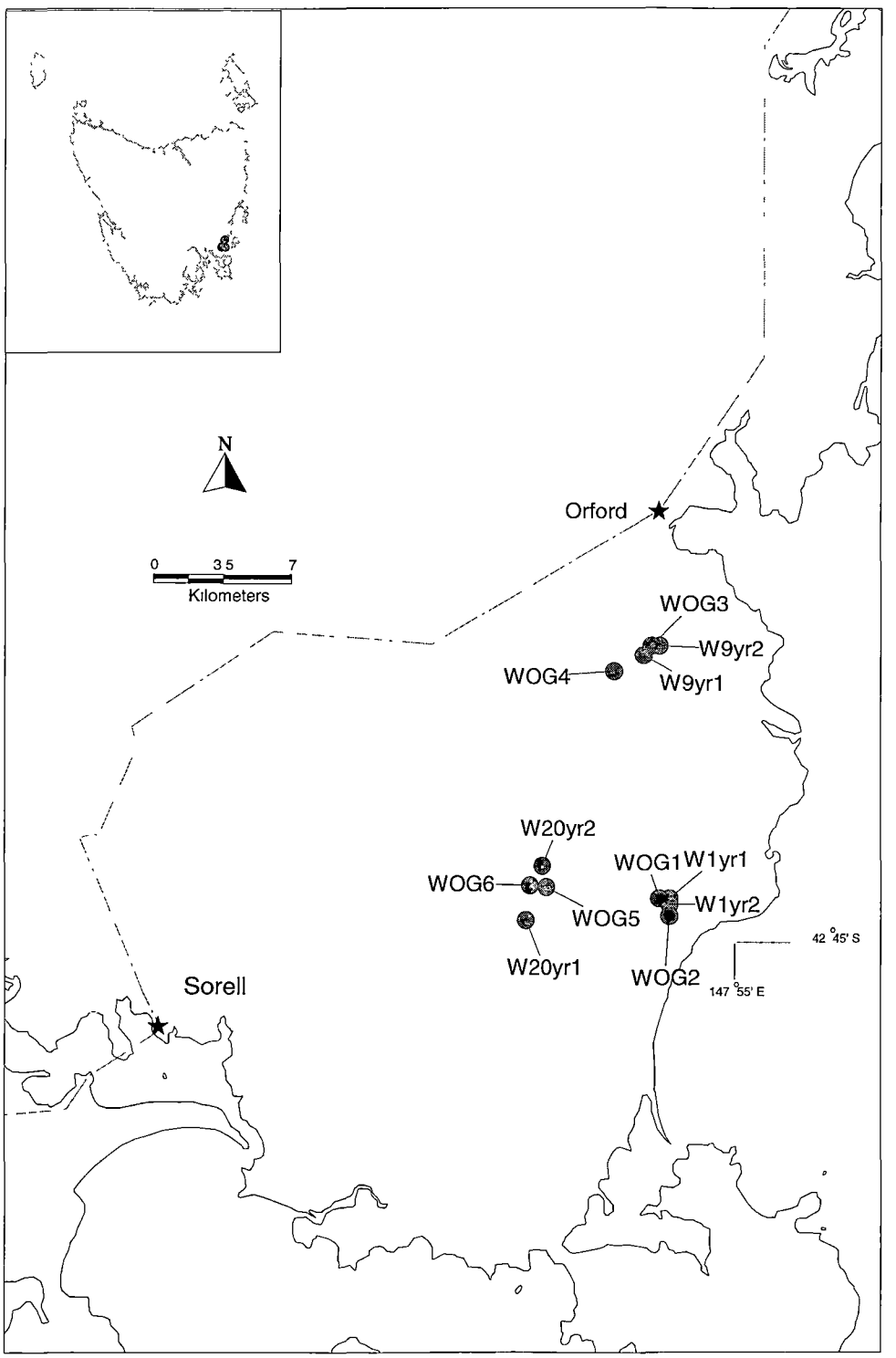


Figure 4.26. Geographic location of study sites in Weilangta.

Code	Locality	Age	Regen. class	Map reference	Easting	Northing
W1Y1	Mount Walters	1	Early	Kellevie	571700	5268700
W1Y2	Mount Walters	1	Early	Kellevie	571600	5268300
W9Y1	The Thumbs	9	Intermediate	Orford	570400	5281300
W9Y2	The Thumbs	9	Intermediate	Orford	571200	5281800
W20Y1	Back Run Hills	20	Late	Sandspit	564300	5267600
W20Y2	Back Run Hills	20	Late	Sandspit	565200	5270400
WOG1	Mount Walters	OG	Old-growth	Kellevie	571200	5268700
WOG2	Mount Walters	OG	Old-growth	Kellevie	571700	5267800
WOG3	The Thumbs	OG	Old-growth	Orford	570800	5281800
WOG4	The Thumbs	OG	Old-growth	Orford	568900	5280500
WOG5	Back Run Hills	OG	Old-growth	Sandspit	565400	5269300
WOG6	Back Run Hills	OG	Old-growth	Sandspit	564500	5269400

Table 4.13. Site location details for Weilangta: site codes, locality, age (years since clearfell; OG = old-growth control), regen. class = regeneration classification as used in the text of this thesis, and location data (TASMAP 1:25000 reference).

Site	S(°)	Asp.	Elv.	pH	ST
W1Y1	12	ESE	360	5	SL
W1Y2	21	S	280	6	SL
W9Y1	12	S	410	6	CL
W9Y2	9	SW	450	6	SL
W20Y1	11	NNE	310	6	SaCL
W20Y2	13	NNE	290	6	SCL
WOG1	12	SE	370	6	SaL
WOG2	20	WSW	230	6	SL
WOG3	19	NW	400	6	SCL
WOG4	25	SW	250	6	SCL
WOG5	12	W	270	6	SCL
WOG6	14	WSW	240	6.5	SCL

Table 4.14. Environmental parameters for sites in Weilangta. S = slope; Asp = aspect; Elv = elevation (m.a.s.l.); pH = pH of surface horizon; ST = soil texture (SaL = sandy loam; SL = silty loam; SaCL = sandy clay loam, SCL = silty clay loam; CL = clay loam).

Forest class	Vegetation description
Early (1 yr)	Little vascular vegetation, except for a few widely scattered clumps of *Goodenia ovata* and *Pteridium esculentum*. There is a considerable amount of logs and charred log debris.
Intermediate (9 yr)	Vegetation consists of an oversorey of scattered *Eucalyptus obliqua* and *Eucalyptus delegatensis*, from 4-13m in height, an understorey of *Acacia dealbata*, *Leptospermum scoparium*, and *Helichrysum apiculatum*, with scattered clumps of *Lomandra longifolia*.
Late (25 yr)	*Eucalyptus pulchella* and *Eucalyptus obliqua*, 10-16m in height, dominate the overstorey, with occasional *Eucalyptus globulus*. The understorey is comprised of scattered *Banksia marginata*, *Leptospermum scoparium*, *Callistemon pallidus* and *Cyathodes glauca*, 3-7m in height, with a ground layer of *Coprosma quadrifida* and *Astroloma humifusum*.
Old-Growth	Vegetation is characterized by scattered *Eucalyptus pulchella* and *Eucalyptus obliqua*, from 35-60m in height, with occasional *Eucalyptus globulus*, a midstorey of *Callistemon pallidus* and *Leptospermum scoparium*, 6-14m in height, with *Helichrysum apiculatum*, *Coprosma quadrifida*, and *Lomandra longifolia* in the groundlayer.

Table 4.15. Typical vegetation at sites within each forest age class in Weilangta.

Fig. 4. 27. Early regrowth site in Weilangta, site W1yr1.

Fig. 4. 28. Intermediate regrowth site in Weilangta, site W9yr2.

Fig. 4.29. Old-growth control site in Weilangta, site WOG4.

Chapter Five

Sampling and Identification

Sites were sampled to produce species lists of carabids and other ground-dwelling Coleoptera. The sampling and identification protocols used to produce these lists are important for two reasons. Firstly, in providing the data for this thesis and secondly, because 'easily sampled' and 'readily identifiable' are consistently stated as crucial *a priori* criteria for selecting potential indicator taxon/taxa (see Chapter Two).

This chapter is divided into two sections:

Section One: Sampling:

- outlines the sampling protocol used
- explores the pitfalls of using pitfalls
- compares the use of pitfall traps against the criteria for sampling required of indicator species

Section Two: Identification:

- examines the issue of taxonomic resolution
- outlines the identification process
- discusses the problems involved in the identification process

5.1. SAMPLING

Sampling procedures play an important part in studies of population and community ecology and it is desirable that they reflect the sampling objectives. The sampling objective was to provide a species assay (inventory) for each site in order to compare carabid diversity and distribution patterns with those of other ground-dwelling Coleoptera.

SAMPLING PROTOCOL

All Coleoptera were sampled using pitfall trapping.

Sampling target

This study only sampled the ground fauna - that is, ground-dwelling Coleoptera. It is likely that some species were under represented or not sampled at all. Subterranean and cryptic species, highly vagile flying beetles and arboreal and foliage beetles are not well censused by pitfall trapping. However, no sampling method evenly covers all habitats or habitat patches that occur at a site and most sample a restricted component of the target group species present (Hammond 1994). Further, according to Martikainen *et al.* (1998) a true total number is an abstraction and the number of species actually present at any site is likely to be higher than can be detected, regardless of sampling effort (Siitonen 1994; Martikainen *et al.* 1998).

The purpose of sampling is to obtain a fixed proportion of target group species at a site, to do it reliably, with some precision and with a minimum of effort (Hammond 1994). Pitfall trapping reliably samples a major, albeit incomplete, component of the beetle fauna. And, to test the validity of biodiversity and ecological indicators, it is only necessary that they reflect aspects of at least a subset of other taxa (Kremen *et al.* 1993; Vane-Wright *et al.* 1994; McGeoch 1998).

Pitfall traps

Each trap comprised an unbaited plastic drinking cup, 7 cm in diameter, 10 cm in depth, inserted into a PVC sleeve in order to minimise ground disturbance during subsequent collections. A 12 cm diameter plastic lid was supported a few centimetres above each trap in order to keep out rain and discourage predation by

birds and mammals. Covers may also help in attracting invertebrates seeking covering objects (Clark and Blom 1992).

Undiluted ethylene glycol was used as a preservative. There are several good reasons for using ethylene glycol (commercial antifreeze) in pitfall traps. One is that it extends the length of time possible between sampling periods. Ethylene glycol because of its relatively slow rate of evaporation and preservative characteristics has successfully been used as a preservative in long-term unattended traps (Clark and Blom 1992). Secondly, although it can be bought in large containers making it cheaper, it is also readily available in a variety of outlets everywhere, handy if the supply has to be supplemented in the field. And thirdly, it overcomes the problems of retaining efficiency: for example, the rate of escape of beetles from dry traps, winged species in particular, and the probability of smaller beetles being eaten by larger ones.

Trap placement

Traps were laid out along a transect or grid. Prior to selecting the placement of the transect, a visual inspection of the site was made so that the placement of the traps took in a representative sample of the range of habitat structures. Since individual species are likely to have preferences for particular microhabitats, a representative example of habitat type covering the range of microhabitats should produce a representative example of the range of species utilising that habitat. Traps were installed with the opening flush with or slightly below the soil surface; on a slightly raised bed so as to avoid flooding due to rain.

Sampling period

The total sampling period was a minimum one year (365 trap days) at all sites with samples collected on an approximately six-weekly cycle.

Sampling effort

Although the sampling effort (number of traps per site) was not identical for all sites, it was, with the exception of sites in the Picton, identical at all sites within each site-group. However, the number of pitfall traps effectively active during the whole-year cycles were in practice less than the number stated, due to several kinds of uncontrollable factors, such as floods, fires, and faunal (native and exotic, i.e. sheep, wombat and possum) damage. The sampling dates and total sampling effort for each of the site-groups were as follows:

Eastern Tiers: Twelve traps were arranged in two rows of six, the traps were placed at 10 m intervals and the rows were 10 m apart. Sampling took place from 18th November 1995 to 17th February 1997.

Grasslands: Twelve traps were arranged in two rows of six, the traps were placed at 10 m intervals and the rows were 10 m apart. Sampling took place from 18th March 1995 to 18th March 1996 at twelve sites and from 1st November 1995 to 1st November 1996 at the other eleven sites.

Coastal Sand dunes: Seven traps were placed along a transect beginning at the initial vegetation just above the high tide zone, traversing the first dune and ending at the first dune swale. Transects were oriented perpendicular to the beach and traps were sited in approximately the same position relative to the dune vegetation structure at each site. Sampling took place from 18th June 1994 to 19 June 1995.

Picton Valley: Traps were arranged at 10m intervals along a 50 metre transect. Five sites had 1 trap at each interval along the transect (6 traps), and five sites had 1 trap at four locations and three traps at each of two other locations (10 traps). Sampling took place from July 18th 1994 to July 18th 1995.

Weilangta State Forest: Traps were set out at 10m intervals along a 50 transect. Sites in the Weilangta dry sclerophyll forests had one pitfall trap at each ten metre interval, a total of six traps at each site. Sampling took place from July 18th 1994 to July 18th 1995.

THE PITFALLS

A pitfall trap is a passive catching device, and capture results from the activity of the target organisms. A wide variety of physical and biological factors are likely to influence the quantity and composition of the catches, including trap size, type and number, placement and climatic factors, the behavioural features of the sample population and the distribution in time and space of the trap itself.

The trap itself

Size

The size of the trap used may influence the number of individuals caught. Significant correlations between the total catch and the available trapping area were found in a study of Southern Mountain arthropods (Turner 1962). Using pitfall

traps of two different diameters, Turner found that the the ratio of catch between the two types of traps (1.58:1) matched the ratio of the trap circumference (1.52:1). Similarly, following a study of tenebrionid beetles in Arizona, Ahearn 1971) found that the catch size was closely related to the amount of exposed trapping surface within each sampling area. He concluded that the efficiency of a pitfall method is directly related to both the individual trap circumference and the total trap density, each of which represents a measure of the available trapping area.

The size of traps may also influence the size of beetle caught. Luff (1975) tested the capture efficiency of 6 types of (dry) trap and found that small traps caught small beetles most efficiently and large traps caught more large beetles although the largest species was caught poorly by nearly all traps. However, since the traps in this experiment were dry, retaining efficiency was a factor. Of most importance is the fact that, regardless of size, all traps caught around 75% of beetles that contacted their perimeter (Luff 1975).

Type

The type of trap may also influence retaining efficiency, particularly where traps are used dry. Using dry traps, Luff (1975) found that the rate at which beetles escape from glass traps was negligible, approximately 4% per day from plastic and 10% from metal. Large beetles most easily escaped from small traps (by straddling the pot and walking up both sides with their legs spread wide) and small beetles escaped most easily from large traps (by being lighter and able to utilise scratches or other imperfections as claw holds) (Luff 1975).

Number

Estimates of adequate numbers vary. Refseth 1980) considers 10-20 traps sufficient for quantitative analysis, whereas Thiele (1977) suggests that 10 traps will give reproducible figures for relative frequency, and over longer trapping periods (8 months or more), 5-10 traps will suffice to show up all dominant species. While an increased number of traps may result in an increase in the number of species caught, the increase is often in the form of species represented by one or a few individuals (Thiele 1977). Further, these species are more likely to be vagrants rather than residents (Hammond 1994), since the likelihood of capture is probably proportional to the length of time spent in the area (Ahearn 1971). As sample efforts change, the representation of common and less common species and also vagrants is likely to vary. If, unlike resident species, vagrant species generally accumulate at approximately the same rate, their influence on inter-site richness may

be marked with increasing sampling effort at sites (Hammond 1994). It is also important to be aware of the possibility of depleting the fauna in a long term study.

Trap capture efficiency

The use of pitfall traps to compare species activities relies on the assumption that every species has the same chance of being captured. But species size, trap placement, microhabitat preferences and behavioural features of individual species and disturbance may all influence the capture rate.

Species size

Greenslade 1964) demonstrated that the larger the beetle, the greater its chance of being caught, attributing this to an increase in the rate of trap encounters since faster-moving larger beetles cover greater distances. If larger beetles range over a greater areas, their probability of capture in pitfalls will be greater (Spence and Niemelä 1994).

Seasonality

Trap captures will also be influenced by seasonality, since many species are only in the 'catchable' adult stage for a few months each year, and this period will vary for different species. This problem can be overcome by continuous pitfall trapping for a minimum one year period (Thiele 1977; Michaels and McQuillan 1995).

Trap placement

Trap placement is important. The physical structure of the habitat around the trap and the height of the trap mouth within the vegetation layer can alter the numbers and types of carabid beetles caught. Some carabids aggregate (Thiele 1977; Niemelä *et al.* 1992) and localised high densities of individuals may result in higher catches, particularly when trap placement coincides with individual species particular microhabitat preferences. For example, Penny (1966) trapped more individuals of the common English carabid *Nebria brevicollis* in pitfall traps located under trees. Rickard and Haverfield (1965) found that one species of tenebrionid, *Stenomorpha puncticollis* exhibited a definite preference for shrub cover while a related species, *Pelecyphorus densicollis*, preferred more open ground.

The placement of the trap lip at either the soil surface or the litter surface may also affect trap catches, possibly by influencing trap encounters (Greenslade1964). Halsall and Wratten (1988) found that the trap "capture efficiency" (the proportion

of encounters resulting in capture) depended on whether the trap was lipped or not, with increased capture rates where the lip was below the substrate. But this did not apply to all species. They considered that some species were more susceptible to capture, and this was related to the species' differing ability to perceive the trap edge. Some species simply skirted around the edges while others "investigated" by hanging over the edge before retreating. However, the traps used in their experiments were dry and also made from polystyrene foam and this probably decreased their capture efficiency, by allowing the possibility of escape in the first instance, and by providing claw holds for the beetles in the second. It is noteworthy that, although capture efficiency was low for all species, no species avoided capture consistently or totally.

During a study on the effect of habitat structure on pitfall trap efficiency in grasslands Greenslade (1964) found trap captures were lower in dense grass and suggested that pitfall trap captures may be effected by habitat density. Following a survey of 23 grassland sites and an experimental *Themeda* plot, manipulated to create three levels of habitat structure to examine the effect of grassland type on slugs and crickets, Melbourne *et al.* (1997) found that habitat structure affected pitfall trap efficiency for crickets but not for slugs. However, they concluded that the results from the experiment indicated that the survey results still reflected true patterns of abundance for crickets and slugs.

The digging in effect

Catches immediately after a pitfall trap is placed in position are commonly found to be higher than those subsequently achieved. This was termed the "digging in effect" by Greenslade (1973). One explanation is that the local disturbance caused by the trap placement attracts some species. In an experiment to dig out the digging in effect, Digweed *et al.* (1995) demonstrated that sustained disturbance increased pitfall catches of carabids, although individual species responded differently. Disturbance led to a slight increase in the number of species captured, a marked increase in the number of individuals of some species captured and to some differences in the structure of the carabid assemblages. Although species richness was higher, evenness (Pielou's J') and species diversity (as measured by the Shannon-Wiener index) was lower in the disturbed trapping stations relative to the permanent (undisturbed). The number and catch of rare (<1% of catch) species was similar among trapping methods. However, it is interesting to note that carabid catches in both treatments decreased over time. Although this was attributed to a seasonally influenced decrease in activity for most carabid species (Digweed *et al.*

1995), it is also possible that the decreased subsequent catches were due to the initial rapid depletion of the population locally.

Climatic factors

Rainfall and temperature may have a marked effect on pitfall captures. The activity of individual species may vary depending on the temperature on particular dates (Spence and Niemelä 1994). Assessing the populations of two species of carabid beetles, Briggs (1960) noted that increased captures were often associated with recent rain and higher temperatures. He found a positive relationship between numbers trapped and temperature, although results were inconclusive for rainfall. After a period of sustained high temperatures and low precipitation reduced the numbers of tenebrionids trapped at South Mountain Park in Arizona, Ahearn (1971) concluded that rainfall and ambient temperature significantly influenced the size of the catch.

SAMPLING CRITERIA FOR INDICATORS

Ease of sampling is one criterion on every proposed list of desirable attributes which indicator taxa or groups should possess (see Chapter Two). Following is a comparison of pitfall trapping against the criteria for sampling required of indicator species.

Simple and Cheap

Pitfall traps consist of an open-mouthed collecting vessel, any one of many different designs, buried in the ground with the opening level with the soil surface such that cursorial animals fall into the trap and are unable to escape. You don't get much simpler than a container in a hole in the ground. In addition, pitfall traps are cheap and very cost effective; and can provide an impressive array of data (Gist and Crossley 1973; Clark and Blom 1992; Spence and Niemelä 1994). A wide variety of containers such as disposable food or drink containers can be used, including such common household items as used metal coffee cans and plastic drink jugs which would normally be discarded (Clark and Blom 1992).

Practical to use

Pitfall traps are easily transported and serviced and as a passive device, they can be left unattended for lengthy periods. Traps may be operated empty for live catches or with preserving fluid or bait. For example, Clark and Blom (1992) collected

well preserved samples from traps that had been out in the Central Desert of Baja California for up to three years, and considered the major advantage of pitfall traps was the short time (5 minutes or less) required to service the traps. The setting and servicing of pitfall traps requires little labor (Halsall and Wratten 1988; Spence and Niemelä 1994) and no special expertise, other than a minor talent for digging holes.

Defined sampling methods

The sampling method is well defined, a container in the ground: the main differences are that variously sized cups (containers), filled with various fluids are left in the hole in the ground for varying periods of time. Since Barber (1931) first reported the use of empty tin cans as insect traps, pitfall traps have been the most widely and commonly used method of sampling and monitoring ground-dwelling arthropods such as ants, spiders, collembola, centipedes and beetles, especially carabid beetles (Briggs 1960; Greenslade 1964; Luff 1975; Southwood 1978; Dennison and Hodkinson 1984; Niemelä *et al.* 1992; Desender 1996; Michaels and Mendel 1998).

SUMMARY

One of the more important practical aspects involved in the choice of sampling methods is the number and dimensions of samples needed to obtain reliable results; since the sampling method must provide sufficient data (ie. an adequate number of target group species) to be informative (ie. reflect real differences between sites or areas). In addition, the sampling method has to be sufficient to eliminate or minimise bias (Hammond 1994). Samples of relatively modest dimensions should be be sufficient to indicate species richness relationships between sites, particularly when activity-based trapping methods are used (Thiele 1977; Hammond 1994). The minimum number of traps used at any one site (6) was within the range considered to provide sufficient informative data (Thiele 1977). The dimensions of the traps were consistent with those used in many current studies, and traps were installed so as to minimise possible bias due to placement. In addition, the use of ethylene glycol increased trap capture efficiency by eliminating the possibility of escape.

The contribution that activity makes to capture size is impossible to evaluate. Pitfall traps are inherently passive, so increased activity by beetles is likely to result in increased catches and conversely, decreased activity in decreased catches. The total capture of ground dwelling arthropods in pitfall traps will be dependent on the

number of individuals crossing the sampling area with the likelihood of capture probably proportional to the length of time spent in the area (Ahearn 1971). The number of individuals trapped will be largely determined by the species surface activity and its population density. Temperature, moisture and other weather conditions and habitat characteristics will all influence activity. Traps were left in place for a minimum one year period. This trapping period overcomes many of the problems associated with activity, since it increases the probability of trap encounters and minimises bias caused by individual species behavioural features, climatic, seasonal and temporal variations in trap efficiency.

As with most sampling methods for living and mobile organisms, pitfall traps do not sample all species in the same way because all species do not behave in the same way (Dufrene and Legendre 1997). However, although pitfall trap data is considered a measure of activity rather than absolute abundance (Digweed *et al.* 1995), they are considered to provide a reasonably good indication of a species quantitative importance (Loreau 1992) and relative abundance when sampling is conducted over long periods, such as a year (Baars 1979; Spence and Niemelä 1994) and appropriate to compare relative abundances of the same species in several sites (Dufrene and Legendre 1997).

Moreover, although one of the main criticisms of pitfall traps is that they measure activity rather than absolute abundance, samples obtained by activity-based trapping methods effectively sample from a larger scale that that evident from the simple distribution of sampling sites (Hammond 1994). However, this characteristic may make them especially appropriate for species richness assays of individual sites as the population (in the statistical sense) sampled more or less equates with the 'target' population (see also Eberhardt and Thomas 1991), i.e. that of the whole site (Hammond1994).

Detailed studies of their efficiency have generally concluded that although pitfall trapping may have limited value in estimations of absolute population size they collect arthropods in numbers suitable for rigorous statistical analyses (Spence and Niemelä 1994); are very useful for measuring population changes in distribution, dispersal and density (Briggs 1960; Greenslade 1964; Ahearn 1971; Luff 1975) and are considered the best method for comparing arthropod assemblages concurrently over a range of sites (Rushton *et al.* 1989; Niemelä *et al.* 1993; Spence and Niemelä 1994). Indeed, where the objective is qualitative inventory and comparison of assemblages over large geographical areas, pitfall trapping is the only realistic alternative presently available (Niemelä *et al.* 1990, Niemelä *et al.* 1993a; Spence

and Niemelä 1994). Pitfall traps allow carabids (and other arthropods) to be easily sampled and therefore fit the sampling criteria for selection indicator species.

5.2 IDENTIFICATION

The identification of species provides the foundation for any ecological evaluation or assessment for both biodiversity inventory and monitoring purposes. Identification is also an integral part of conservation and management because it is the basic tool in communicating or extracting information about species. The identification of taxa thus played an integral role in the formulation of this thesis.

TAXONOMIC RESOLUTION

The beginning of wisdom is calling things by their right name.
Krishtalka, China (Evans and Bellamy 1996 p. 12)

Identification is the process by which each of the individual organisms sampled during this study were assigned names. Names are the unique labelling code by which individual organisms are allocated independent but related identities within the overall taxonomic classification of the vast range of organic diversity (Hawksworth and Ritchie 1993). The identification process places individuals into previously established taxonomic classes using deductive procedures (Mayr 1969) based on resemblances and differences (Mayr 1969; Abercrombie *et al.* 1973). A taxonomic class is a category designating rank or level in a classification system, the members of which are all the taxa assigned a given rank (Mayr 1969). I.e., the species category is a class, the members of which are the species taxa. All individuals in this thesis were identified to species level.

Ecological surveys and ecologists depend upon accurate identification. Species lists provide the baseline data for biodiversity inventories, since any assessment of biodiversity includes an estimate of species richness. Establishment of reserves is dependent upon data about identified organisms (Prance 1995), since reserve establishment is targeted at areas with high "species richness". The presence of threatened, endangered or rare species and the evaluation of representativeness are all assessed on the distribution of taxa at the species level, as are biogeographically important areas such as centres of endemism, refugias, and "hotspots" of diversity (McNeely *et al.* 1989). Species are also the focus of environmental impact assessment and monitoring, with the detection and measurement of changes in the biota of an ecosystem, such as variations in the relative abundance of species, reliant on the identification of and use of species level taxa (Soulé and Kohm 1989; Hawkesworth 1994; McLaren *et al.* 1998). Successful management for sustainable

conservation therefore relies on an understanding of the biology and ecology of species. This understanding depends on reliable and accurate knowledge that can only be achieved through accurate identification.

IDENTIFICATION PROTOCOL

Identification begins with individuals (Mayr 1969).

Individual specimens were identified by the author, with some assistance from Dr Peter McQuillan, and relevant specialists. All specimens were sorted on the basis of external morphology to species level and the frequency of these recognised taxonomic units (referred to hereafter as species for convenience) were recorded.

Identification process

The identification process took several steps.

Step 1

The contents of the pitfall traps were rough-sorted. The mix of antifreeze and specimens was poured into a wide dish and all Coleoptera recognised by the naked eye or with the aid of a magi-lamp were extracted. The remaining contents were then examined carefully under a microscope and all other individual specimens identified as Coleoptera were removed. During this process external morphological characters were used to sort and group specimens into morphotypes.

Step 2

Further examination was made of one or more individuals of each final grouping. Keys supplemented by reference to the voucher collection held at the School of Geography and Environmental Studies, University of Tasmania, were used to determine genus and species identification where possible. Dr Peter McQuillan was also consulted whenever necessary to further define this preliminary identification.

The first specimen encountered from each new species was allocated a name; either a species name or a morphospecies pseudonym where it was not possible to accurately assign a taxonomic name. A label was attached to the specimen. This label recorded the combination of Family, Genus and/or species name that had been assigned after this examination. Another label recording site code, site location and the date of collection was also affixed. These specimens were then lodged in the

voucher collection held at the School of Geography and Environmental Studies University of Tasmania. Where reference to the voucher collection produced a match (ie. a voucher specimen of the species being examined was already present in the voucher collection), one of two actions was taken.

(a) If the specimen occurred in a different habitat type from the voucher specimen (i.e. the first specimen had been collected from dry sclerophyll forest and the subsequent specimen in grasslands), a label recording the predetermined identification details (Family, Genus and/or species name) was attached and the new specimen was lodged in the voucher collection.

(b) If the specimen occurred in the same habitat type as the voucher specimen, it was assigned its name and the data recorded in the inventory.

Step 3

All other individual specimens which had been grouped together on the basis of similar external morphology (i.e. shape, size, colour or a special and noticeable characteristic) to the identified specimen were reexamined with the microscope to determine if they possessed the same morphological characters or combination of characters as the named specimen from the group. Matching individuals were assigned the temporary name of each predetermined species. Non-matching individuals were reexamined, and step two was repeated until a species name or morphospecies pseudonymn could be allocated. Individual specimens in each allocated name group were then counted and the count recorded in an inventory.

Step 4

To confirm identifications, specimens were compared with verified species where-ever possible. Temporarily named specimens were first taken to the Tasmanian Department of Primary Industries and Fisheries (DPIF) museum reference collection, and compared to the voucher specimens held there. Where a match was achieved, the first allocated name was confirmed. Where comparisons revealed differences in certain or any characteristics and therefore led to doubts of the accuracy of the name allocation, the specimen was compared to other specimens within the determined genus. Where a match was achieved, the name of the DPIF voucher specimen was allocated and a second temporary label was affixed to the reference specimen. The first label still remained with the specimen. Confirmation of the first determination and/or changes to names allocated during the preliminary

identification were recorded and the data in the inventory corrected where necessary. The specimen was then relodged in the voucher collection held at the School of Geography and Environmental Studies University of Tasmania.

Step 5

Some specimens were forwarded to specialists for (a) confirmation or correction of the temporary identification label; (b) supplementary identification to species level; or (c) confirmation that the species had yet to be described. These specimens were then given a third label. The label recorded the taxonomists determination of Genus and species or morphospecies name where the species had yet to be described. Confirmation or corrections were recorded and the data in the inventory corrected where necessary. The specimen was then relodged in the voucher collection held at the School of Geography and Environmental Studies University of Tasmania.

Components of the identification process

There were three critical components in the identification process:

- keys,
- reference collections and
- specialist input.

Keys

Several keys were used to determine Coleoptera specimens to family and genus level: Lawrence and Britton 1994; Moore 1981; and Mathews 1980, 1982, 1984, 1985, 1987, 1992, 1997. Sloane (1920) was used to determine specimens of Carabidae to species level.

Reference Collections

Reference collections were an essential component of the identification process. Some were consulted constantly: the author's own voucher collection; the voucher collection held at the School of Geography and Environmental Studies University of Tasmania and the Tasmanian Department of Primary Industries and Fisheries (DPIF) museum reference collection at NewTown in Tasmania. The South Australian Museum reference collection in Adelaide and the Australian National

Insect Collection (ANIC), at CSIRO in Canberra were also consulted on several occasions.

Specialist Input

Specialist taxonomists, Dr Barry Moore, Dr John Lawrence and Dr Tom Weir (Carabidae); Dr Andrew Calder (Elateridae) and Dr Zimmerman (Curculionidae) at the Australian National Insect Collection (ANIC), CSIRO, Canberra, Dr Eric Mathews (Tenebrionidae) at the South Australian Museum and Dr George Bornemissza (Lucanidae), a research fellow with CSIRO, resident in Tasmania, examined voucher specimens and provided valuable information and assistance.

PROBLEMS

Promecoderus

The greatest difficulty in the identification process came from the genus *Promecoderus*, which contains several sibling forms in Tasmania. Sibling species are species that are identical in outward appearance or very nearly so, but qualify as species by being reproductively isolated (Allaby 1991). Lack of morphological variation and morphological overlap means sibling species are very hard to positively resolve. For example, in a study of allozyme and morphological variation in tuatara (lizard like reptiles) in New Zealand to establish if *Sphenodon punctatus* is more than one species, although three distinct groups (separated geographically) were distinguished differing at a minimum of 3 of 25 allozyme loci examined, discrete groups could not be identified using morphological characters (Dougherty *et al.* 1990). Moreover, other workers suggested that the taxonomic differences among proposed taxa were no greater than observed within individual colonies (Dougherty *et al.* 1990).

Differences between closely related species usually increase (diverge) when they occur together, in a process called character displacement, which may be morphological or ecological (Allaby 1991). Similar species of carabids are rarely found together, with sibling species generally distributed allopatrically or parapatrically (Hammond 1990). Using the morphological characters from Sloane's (1920) key and a preliminary examination of male genitalia using microscope and SEM (scanning electron microscope), it was determined that the *Promecoderus* species were allopatric, since specimens at sites within a biome exhibited little or no morphological variation. However, the possibility that the presence of sibling forms may have resulted in a slight underestimation of the actual

species number (one or two species) at each study site where *Promecoderus* was present cannot be overlooked.

A species is "what a competent taxonomist says it is" (Stevens 1992)

Morphospecies pseudonyms were used for many species in this study, particularly those in families other than Carabidae. There were three main reasons for this. Firstly, keys for identification of many Tasmanian species in families other than Carabidae were not available. Secondly, many species were not represented in available voucher collections and thirdly, many Tasmanian species in families other than Carabidae have not been formally described. This is particularly true for the Curculionidae. Although there is a key for most Tasmanian Carabidae (Sloane 1920), several undescribed species were recorded during this study.

It has been suggested that the use of the term morphospecies may represent no more than a tacit admission that taxa are unreliably sorted (Hammond 1994). The question of how easily identifiers who are not taxonomists specialising on the group in question are able to reliably sort to species has been addressed by various authors (Cranston and Hillman 1992; Beattie *et al.* 1993; Oliver and Beattie 1993). It is generally considered that where operational procedures for sorting RTU's (recognisable taxonomic unit - a term often used interchangeably with morphospecies) are 'standardised and calibrated' by conventional taxonomic methods, results achieved by "biodiversity technicians" will be good, although this may vary with the group, since some groups are easier to identify than others (Cranston and Hillman 1992; Oliver and Beattie 1993).

Following an invertebrate survey of five sites in north-east Tasmania, a concordance of almost 100% was obtained between Coleoptera RTUs as identified by inexpert project personnel, and species as determined by taxonomists, with site rank order the same (Cranston and Trueman 1997). Oliver and Beattie 1996a,b) demonstrated that morphospecies inventories of beetles generated by non-specialists, sorting using external morphology only and without the use of keys, varied little from species inventories provided by taxonomists. Differences were largely due to problems of identification in the families, Curculionidae and Staphylinidae. However, both morphospecies and species inventories produced identical rankings of forest type based on richness. Moreover, assessment of turnover using ordination produced similar clusters regardless of inventory used.

SUMMARY

For this thesis, every effort was made to ensure that every specimen was accurately sorted. Standard taxonomic methods were followed and specialists believe that species of Coleoptera are relatively well circumscribed, and easy to identify (Minelli 1993). Each species was allocated a Latin binomial wherever possible, but morphospecies were used where necessary. In circumstances where ecological study precedes the taxonomy, the use of morphospecies or RTUs is the only practical solution (Cranston 1990).

Chapter Six

Assessing the indicator potential of carabids for inventory:

Carabids as Biodiversity Indicators

The concept of biodiversity indicators has received considerable attention in the area of conservation biology, particularly in reserve-selection because basic inventory data for most taxa are sparse or non-existent (Raven and Wilson 1992; Vane-Wright *et al.* 1994; New 1995). Representation of all existing biota in a network of protected areas, on a regional, national and even continental scale, has become a fundamental goal of conservation planning (Spellerberg 1992; Troumbis and Dimitrakopoulos 1998). However, there is not the time, resources or expertise to carry out detailed inventories for most taxa before designating protected areas.

If biodiversity is to be inventoried and monitored, measurable indicators of its composition, structure and functioning must be identified (Noss 1990). One suggested approach is to focus on (one or more) selected subsets of biota, i.e. biodiversity indicator taxa, on the assumption this subset will be representative of (at least a part of) the remaining unmeasured diversity and that protection for the subset will protect these uninventoried species.

Indicator test measures

A good biodiversity indicator will have patterns such as species richness and biogeographical dispersal that are common to other taxa at the relevant scale (Kremen *et al.* 1993; McGeoch 1998. Obviously if a taxon is going to be used to indicate patterns for other taxa, evidence that the potential indicator actually reflects significant patterns among other taxa is vital (Pearson 1994). However, to date "patterns observed in indicators reflected in at least a subset of other taxa" has been little tested, particularly at the local scale. Two tests of the indicator potential of a particular taxon/taxa are commonly mentioned in the literature. One test is to examine species counts among sites for evidence of covariation or correlation, and a

second is to look for evidence of geographic coincidence between areas that are species rich for both the proposed indicator taxon and for other taxa (Pearson and Cassola 1992; Cranston and Trueman 1997; Flather *et al.* 1997).

Correlated Species Counts

Several studies have attempted to validate the use of biodiversity indicators to identify areas of high diversity by seeking positive correlations between the species richness of a potential indicator and the richness of other groups for which data is available. Where such comparisons have been made at a coarse spatial scale, across widely divergent habitats or ecosystems, or between groups of organisms which do not share the same or even similar ecological requirements, they have, to date, met with little success (Prendergast *et al.* 1993; Oliver and Beattie 1996b; Flather *et al.* 1997; van Jaarsveld *et al.* 1998). A significant correlation between the patterns of species richness exhibited by an 'indicator group' and the more general patterns , for example the comparison and ranking of sites or regions in terms of their species richness or uniqueness, is also considered sufficient for a group of this type to serve the purposes of biodiversity 'assessment' (Hammond 1994). However, even where such studies have focused at the local scale, they have also failed to find evidence of congruence between different taxa (Cranston and Trueman 1997; Greenslade 1997).

Coincidence of hotspots

Places with the greatest diversity of species are commonly referred to as 'hotspots' (Prendergast *et al.* 1993; Noss and Murphy 1995; Tardiff and DesGranges 1998). The term has also been used to refer to sites with large numbers of endemic species (Myers 1990; Williams *et al.* 1996). In this thesis, the term will refer to species richness hotspots. If species counts for an indicator taxon are strongly correlated with other taxa, then, on average, areas where many species of the indicator taxon occur should also be characterized by high species counts of other taxa (Flather *et al.* 1997). High priority is often given to conserving species rich habitats for well studied taxon such as higher plants, birds or butterflies, on the assumption that hotspots for one taxon may correspond to those for other taxa (ICBP 1992). However, to date, there is little evidence to validate this assumption (Prendergast *et al.* 1993; Lombard *et al.* 1995; van Jaarsveld *et al.* 1998).

Reserve selection

If a chosen subset of biota (biodiversity indicator) does reflect the overall patterns of biodiversity at the local level, then the problems of incomplete and taxonomically biased inventories are moot and there will be no need to survey all biota (Flather *et al.* 1997). If (adequate) representation for the chosen subset of biota in a reserve system also provides representation for the uninventoried biota, then there will be no need to consider them in the process of reserve selection. The critical question is: will a set of reserves providing representation for the indicator species also provide representation for other taxa? The next question is: how do we select the minimum number of areas to conserve the range of diversity, i.e. the maximum number of species? There are a number of possible approaches to the problem of choosing which sites to include in a reserve network.

The Hotspots approach

One area selection approach, is to select the set of sites that are species rich (i.e. hotspots) for carabids. High species richness is typically promoted as a means to evaluate area-based priorities (Vane-Wright *et al.* 1991; Spellerberg 1992; Williams and Humphries 1994), and one of the most commonly advocated quantitative methods of selecting priority areas for conservation is choosing hotspots, areas that individually have the highest set of sites species richness (Prendergast *et al.* 1993; Sisk *et al.* 1994). However, selecting a set of sites that are individually species rich will not necessarily produce a set that is species rich *as a set* (Faith and Walker 1996b), nor will they necessarily represent the range of variation within a region or biome, even for the indicator itself (Faith and Walker 1996b).

The Representative species richness (RSR) approach

A second approach is to select a set of representative sites based on the carabid data. Representativeness is a primary criterion in the selection of reserve areas (Margules and Usher 1981; O'Connor *et al.* 1990), and recent studies have emphasised the need to cover the maximum variation of natural features (i.e. species) in a reserve network (Faith and Walker 1996a). The concept of representativeness is based on the premise that there are a range of ecological units, which can be classified in a "biospatial" hierarchy, such as ecosystems (or land systems), communities, and species (Soulé 1991; Rowe 1996), and which represent the range of variation found within a defined biogeographical zone (Margules *et al.* 1988; O'Connor *et al.* 1990; Soulé 1991; Spellerberg 1992). The assumption is that selecting and reserving a proportional sample of these ecological units, will conserve the range of natural diversity (O'Connor *et al.* 1990; Spellerberg 1992).

A key factor in assessing representativeness has been the classification of ecological components such as flora to identify characteristic communities based on species distribution and associations (Kershaw and Looney 1985; Spellerberg 1992), using multivariate classification techniques such as cluster analysis. Cluster analysis identifies the range of variation in biotic or environmental data by breaking it into discrete and complementary groups or clusters (Belbin 1995a). A representative set of areas therefore would be one that samples every cluster (Faith and Walker 1996b). Although one criticism of selecting sites on species richness is that successive sites might just duplicate the species already preserved in the first priority area (Kirkpatrick 1983), in principle, this should be less likely where sites are chosen from within each cluster, since each cluster represents a unique portion of the overall biotic composition (Belbin 1995a). Selecting the most species rich site within each set of representative areas (cluster) could therefore be expected to add more new species (Faith and Walker 1996b).

Complementarity: the greedy algorithm (GA) approach

Another well established approach for selecting sets of protected areas is based on the principles of 'complementarity' (Pressey *et al.* 1993; Margules *et al.* 1994). 'Complementary-areas' methods build up sets of areas by progressively adding sites with high complementarity values, that is, that contribute the most number of so-far-unrepresented features (i.e. species) (Faith and Walker 1996a). Stepwise (iterative) algorithms are often used to identify complementary areas that represent the feature variation in a minimum or near minimum set of sites, in other words, areas that in combination have the highest species richness (Kirkpatrick 1983; Margules *et al.* 1988; Pressey *et al.* 1993; Csuti *et al.* 1997). They are termed iterative because they proceed stepwise and each successive step takes the features of sites already selected into account. One such algorithm is the "greedy" (richness-based) algorithm (Csuti *et al.* 1997). It begins with the most species rich site and sequentially adds sites that adds the most additional species (i.e. the assemblage which is most complimentary) (Csuti *et al.* 1997).

CHAPTER AIMS

The aims of this chapter are:

- to test whether patterns of diversity of carabids are congruent with the diversity patterns of other selected Coleoptera; and
- to test the assumption that reserve selection based on carabids guarantees protection for other (uninventoried) taxa, specifically other selected Coleoptera.

The first aim is addressed by applying two common test measures of indicators to the carabid and Coleoptera data:

1. Correlated species counts, and

2. coincidence of hotspots.

To assess the extent to protection for carabids might result in the conservation of Coleopteran diversity, three different reserve selection approaches were applied to the carabid data:

1. The Hotspots approach - an approach which selects the complete set of sites individually species rich (i.e. hotspots) for carabids.

2. The Representative species richness (RSR) approach - an approach which selects sites with the highest number of species from clusters containing species assemblages which are most dissimilar, i.e. clusters within a hierarchical classification;

3. The greedy Algorithm (GA) approach - an iterative approach which adds new sites to a reserve system according to their contribution of unreserved features i.e. species.

METHODS

STUDY AREAS

The study sites, which were described in Chapter Four, consist of 51 sites from three biomes: 18 sites in the predominantly dry sclerophyll forest of the Eastern and Fingal Tiers, 23 remnant native grassland sites, and 10 coastal sand dune sites. In the text, the groups of sites from each biome will be referred to as the Eastern Tiers, Grasslands and Sanddunes respectively.

DATA ANALYSIS

Data analyses were applied to carabids, all Coleoptera sampled, including carabids (referred to as All Coleoptera), Coleoptera excluding carabids (referred to hereafter as Coleoptera) and four coleopteran families that comprised >5% of the total coleopteran species richness within a biome, and occurred in all three biomes, namely: Curculionidae, Scarabaeidae, Staphylinidae and Tenebrionidae. The proportions of species in each biome are shown in Figure 6.1.

Indicator test measures

Correlated species counts

To explore correlations within biomes based on species counts, simple regression analysis was used to obtain Pearson's correlation coefficients for pairwise comparisons between carabids, All Coleoptera, Coleoptera, Curculionidae, Scarabaeidae, Staphylinidae and Tenebrionidae. Sites were ranked according to their total species richness for carabids, All Coleoptera, Coleoptera, Curculionidae, Scarabaeidae, Staphylinidae and Tenebrionidae. Correlations in site rank ordering between taxa were explored within biomes and within plant community types (PCTs) in the Eastern Tiers and within grassland community types (GCTs) in the Grasslands. Pearson's rank correlation was used to test for the statistical significance of rank order correlation between taxa within plant community types.

Coincidence of hotspots

Species richness hotspots were defined as those sites that contained at least the mean number of species present in the biome in question plus one standard deviation (SD) (see Table 6.1) (Tardiff and DesGranges 1998). Hotspots for carabids, All Coleoptera, Coleoptera, Curculionidae, Scarabaeidae, Staphylinidae

and Tenebrionidae were identified and compared for evidence of congruence. The proportional overlap of carabid hotspots with hotspots for All Coleoptera, Coleoptera and other Coleopteran taxa was calculated as the total number of overlaps in all three biomes divided by the maximum possible number of overlaps, defined as the smaller number of hotspots for the pair adjusted for sites lacking records for either group.

Reserve selection methods

For each biome, three reserve selection approaches were applied to the carabid data from all sites within the biome . The resulting sets of areas, under each selection approach, were evaluated to assess the level of representation provided for Coleoptera when reserve selection is based on carabid data.

To determine representative groups i.e. groups with similar species composition, sites within each biome were classified using the hierarchical-agglomerative-polythetic clustering method "Flexible UPGMA" from the PATN (Pattern Analysis Package) software package (Belbin 1995b). The input is a similarity matrix derived from the species data based on the rank order of all dissimilarities and the output is a dendrogram displaying the results of clusters of associated objects (sites). The Bray-Curtis similarity co-efficient was used to construct the matrix. Faith and Walker (1996b) suggest that the hierarchical clustering method flexible UPGMA with Bray-Curtis dissimilarities (Belbin 1995b) is likely to produce a hierarchy where differences between clusters do indeed reflect the relative number of species differences. The default group setting, Group Definition (GDEF) in PATN (Belbin 1995b) was used to determine the number of defined groups or clusters.

The abundance of each species caught in each site was transformed to a percentage of the total abundance of all species of the taxon to which it belonged in that site (taxon in this case being coleopteran family), thus converting the data to a proportion between 0 and 1. This transformation corrects for site specific effects influencing the community as a whole (van-Straalen *et al.* 1988), compensates for overall differences in the total numbers of species caught in each site due to unknown differences in activity (Luff 1990; Sanderson 1994; Desender 1996) and such proportional data are simpler to interpret in multivariate analyses (Sanderson 1994).

Hotspots approach

All sites identified as Hotspots for carabid beetles within each biome were selected.

Representative species richness (RSR) approach

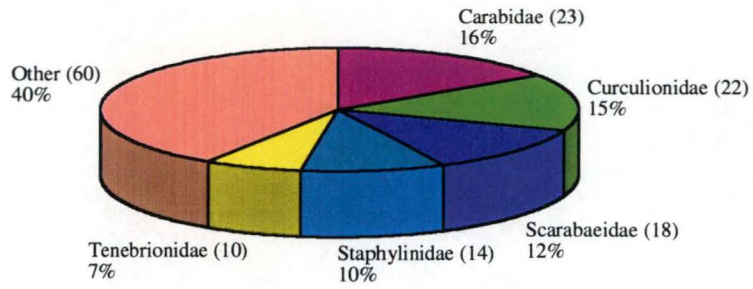
Sites with the highest number of carabid species were selected from each cluster within the hierarchical classification described above. Dendrograms from the UPGMA analysis showing the degree of similarity between sites and the degree of dissimilarity between clusters with regards to the species composition of the carabid beetle fauna for sites in each biome and the clusters recognised by the UPGMA default setting and used in the Representative species richness (RSR) area selection approach are shown, for each biome, in Figures 6.2, 6.3 and 6.4.

Greedy Algorithm (GA) approach

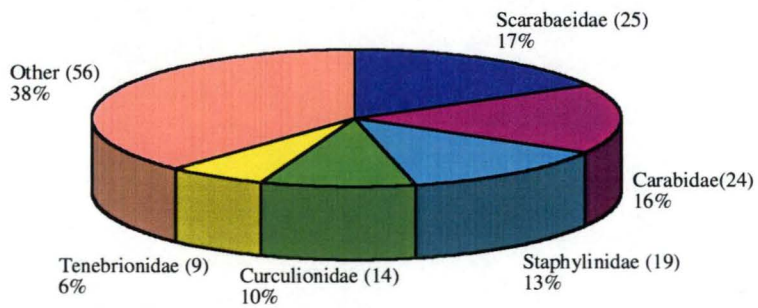
The goal was to identify the minimum number (set) of sites required to represent all carabid species at least once. For these calculations, that were carried out for each of the three biomes, I selected the site with the greatest carabid species richness as the first site (this is by definition a carabid species richness hotspot). For each remaining site, I determined the number of species not already represented in selected sites and chose the site with the greatest species complement (the greatest number of additional species) as the next site; in the event of ties, I selected a site at random, repeating this step until all the species in the biome in question were represented at least once.

The iterative algorithm used is the same one described in Csuti *et al.* 1997). While not necessarily the most efficient of all possible selection procedures, it does tackle complementarity and is very similar to well-tested and powerful algorithms applied elsewhere (see Kirkpatrick 1983; Vane-Wright *et al.* 1991; Williams and Humphries 1994; Balmford *et al.* 1996b; Williams *et al.* 1996).

(a) Eastern Tiers



(b) Grasslands



(c) Sanddunes

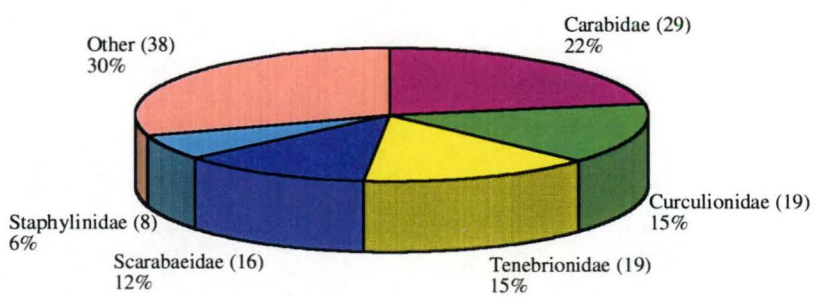


Figure 6.1. The number of species, and the contribution (%) made to the total coleopteran species richness in the biome, for carabids and each of the other four coleopteran families in (a) the Eastern Tiers, (b) the Grasslands and (c) the Sanddunes.

(a) Eastern Tiers

Taxon	Mean spp.	SE	SD	Mean + 1SD
Carabidae	4.9	0.6	2.7	7.6
All Coleoptera	39.8	2.6	11	50.8
Coleoptera	34.8	2.9	12.6	47.4
Curculionidae	7.5	0.8	3.2	10.7
Scarabaeidae	5.9	0.8	3.5	9.4
Staphylinidae	6.2	0.5	2.2	8.4
Tenebrionidae	2.8	0.2	0.9	3.7

(b) Grasslands

Taxon	Mean spp.	SE	SD	Mean + 1SD
Carabidae	7.4	0.8	3.9	11.3
All Coleoptera	40.7	2.9	14	54.7
Coleoptera	33.3	2.3	11	44.3
Curculionidae	6.1	0.5	2.4	8.5
Scarabaeidae	7.5	0.8	3.7	11.2
Staphylinidae	4.3	0.5	2.5	6.8
Tenebrionidae	2.6	0.3	1.4	4

(c) Sanddunes

Taxon	Mean spp.	SE	SD	Mean + 1SD
Carabidae	9.2	0.8	2.4	11.6
All Coleoptera	49.2	2.5	8	57.2
Coleoptera	40	1.9	5.9	45.9
Curculionidae	10.7	0.5	1.6	12.3
Scarabaeidae	5.5	0.5	1.7	7.2
Staphylinidae	3.1	0.9	2.5	5.6
Tenebrionidae	5.9	0.8	2.5	8.4

Table 6.1. Untransformed mean number of species (Mean spp.), standard error (SE) and standard deviation (SD) for Carabidae, All Coleoptera, Coleoptera, Curculionidae, Scarabaeidae, Staphylinidae and Tenebrionidae for each biome.

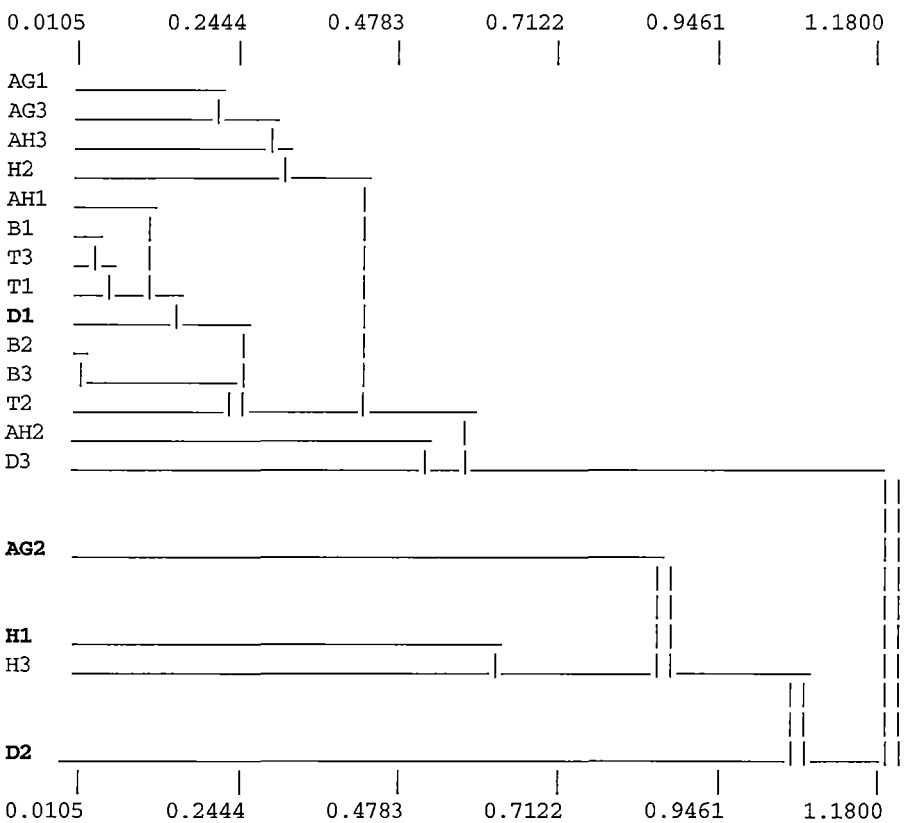


Figure 6.2. Dendrogram from UPGMA analysis, based on the species composition of the carabid beetle fauna for sites in the Eastern Tiers, showing: the clusters used in the Representative species richness (RSR) area selection approach; the degree of similarity between sites (|) within the four clusters recognised by the UPGMA default setting; and, the degree of dissimilarity between clusters (||). The site selected (the most species rich site) within each cluster is indicated in bold. Details relating to the site codes used here are given in Chapter Four, Table 4.1.

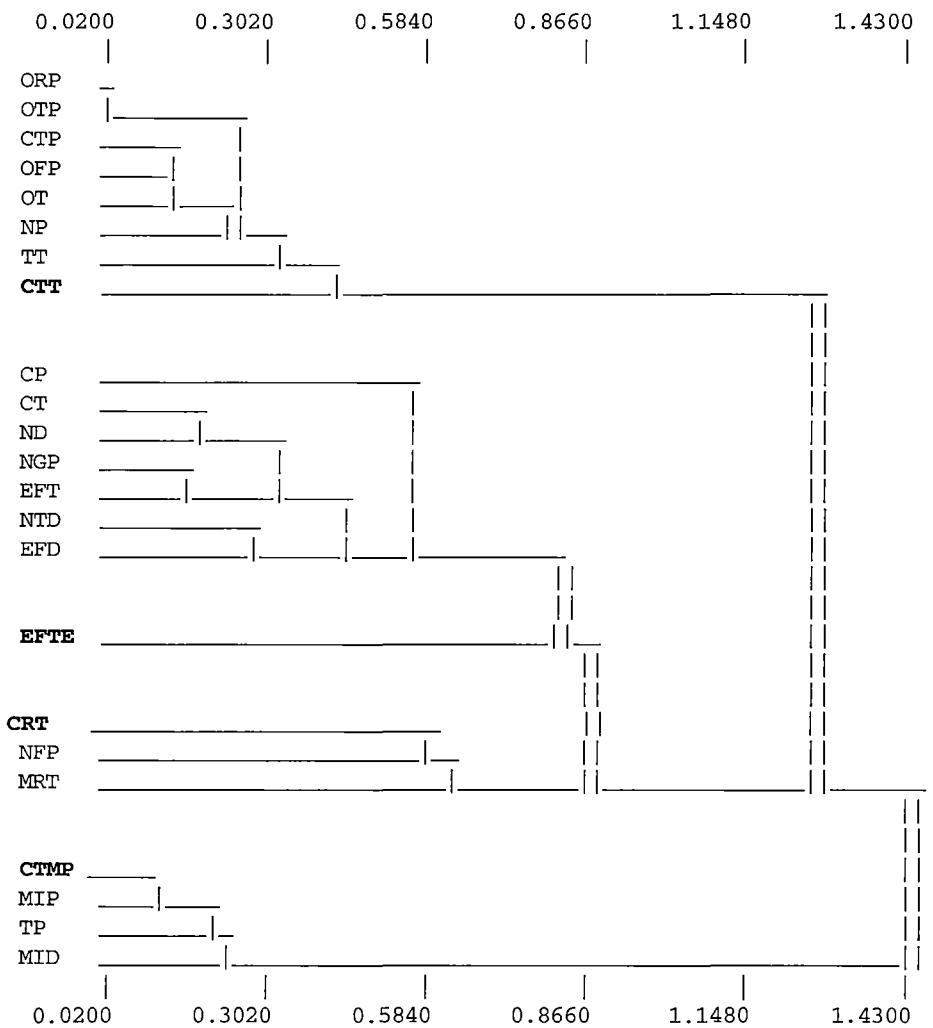


Figure 6.3. Dendrogram from UPGMA analysis, based on the species composition of the carabid beetle fauna for sites in the Grasslands, showing: the clusters used in the Representative species richness (RSR) area selection approach; the degree of similarity between sites (|) within the four clusters recognised by the UPGMA default setting; and, the degree of dissimilarity between clusters (||). The site selected (the most species rich site) within each cluster is indicated in bold. Details relating to the site codes used here are given in Chapter Four, Table 4.4.

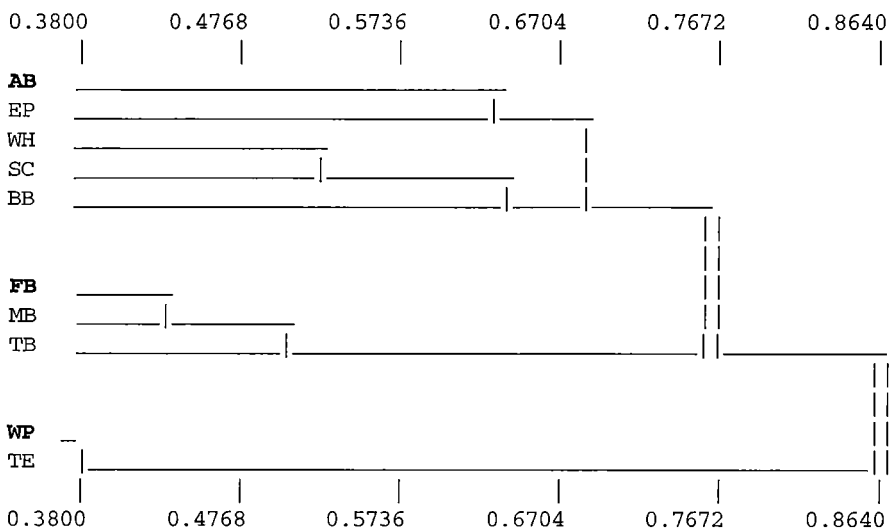


Figure 6.4. Dendrogram from UPGMA analysis, based on the species composition of the carabid beetle fauna for sites in the Sanddunes, showing: the clusters used in the Representative species richness (RSR) area selection approach; the degree of similarity between sites (|) within the four clusters recognised by the UPGMA default setting; and, the degree of dissimilarity between clusters (||). The site selected (the most species rich site) within each cluster is indicated in bold. Details relating to the site codes used here are given in Chapter Four, Table 4.7.

RESULTS

INDICATOR TEST MEASURES

Correlated species counts

There were significant positive correlations in total species counts between carabids and All Coleoptera and between carabids and Coleoptera in all three biomes, although the strengths of the correlations varied (Table 6.2). Correlations were weakest in the Eastern Tiers (r^2=0.24, p = 0.04, n = 18 sites), and strongest in the Sanddunes (r^2 =0.82, p = 0.0003, n = 10 sites). There were also significant positive correlations between carabids and curculionids and carabids and scarabaeids in the Eastern Tiers and the Grasslands but not in the Sanddunes. Moreover, although correlations between carabid species counts and species counts for staphylinids and tenebrionids were not significant, there was evidence of some, albeit generally weak, correlation.

Pearson's rank correlation showed there was no significant correlation amongst site rankings between carabids, All Coleoptera, Coleoptera and the four coleopteran families in the Eastern Tiers (Table 6.2). Each taxon placed the 18 sites in a different rank order (Table 6.3). In the Grasslands (Table 6.4), there was a significant correlation amongst site rank order between carabids, All Coleoptera, Coleoptera, Curculionidae and Scarabaeidae (Table 6.2). Although the remaining taxa ranked the same site as most diverse, there was little agreement past this point. In the Sanddunes (Table 6.5), there was a significant correlation amongst site rank order between carabids, All Coleoptera and Coleoptera. Staphylinidae placed the same site in the first ranking position, as the most diverse site but again there was little agreement on site ranking with carabids and other taxa.

Site rankings on species richness within plant community types (PCTs) in the Eastern Tiers are shown in Table 6.6. It is immediately apparent, that although there were many points of agreement, correlations amongst site rankings in the Eastern Tiers depended on the plant community type and on the taxon. However, site rankings for All Coleoptera and Coleoptera concurred with those for Carabidae in four out of the six PCTs. Site rankings for Scarabaeidae also concurred with Carabidae in four out of the six plant community types, but not the same four. Interestingly, although there was no correlation between carabid site rankings and those of any other taxa in the heathy *Eucalyptus amygdalina* sites (AH1-3),

rankings concurred for all other taxon except Curculionidae. Site rank order based on Staphylinidae species richness concurred with carabid site rankings in five of the six PCTs and site rankings for tenebrionids concurred with carabids in three of five PCTs (tenebrionids were absent from the wet Heath sites (H1-3)). There was no agreement on site rankings between Carabidae and Curculionidae in any of the plant community types. Surprisingly, although there was agreement between site rankings for carabids, staphylinids, scarabaeids and tenebrionids in the *Eucalyptus tenuiramis* sites (T1-3), both All Coleoptera and Coleoptera placed the sites in a different rank order, although the same site was ranked as the most diverse.

There were fewer points of agreement amongst site rankings within grassland community types (GCTs) in the Grasslands (Table 6.7). Site rankings for Carabidae and All Coleoptera concurred in only three out of six GCTs, and in only one GCT for Carabidae and Coleoptera. Site rankings for Carabidae concurred with Curculionidae, Staphylinidae and Tenebrionidae in only two GCTs; Scarabaeidae in only one. In grassland community T1, site rankings for carabids agreed with site rankings for all other taxon except Scarabaeids, although since there were only two sites in this GCT and both ranked equally for carabids, agreement on site ranking depended on which site was ranked first. There was no correlation amongst site rankings for Carabidae and all other taxon in grassland community's T2 and T3 and agreement only between Carabidae and Curculionidae site rank order in grassland community T5. Although there was no agreement on overall site rankings between carabids, curculionids, scarabaeids and staphylinids in grassland community T6, all taxon ranked the same first and second sites.

Coincidence of hotspots

Five sites were identified as carabid hotspots and three sites were identified as hotspots for All Coleoptera in both the Eastern Tiers (Fig. 6.5) and the Grasslands (Fig. 6.6). One site was identified as a hotspot for each in the Sanddunes (Fig 6.7). A qualitative visual inspection of Figures 6.5 to 6.7 reveals that while many hotspots for All Coleoptera overlapped with carabid hotspots, and there was some hotspot overlap in all three biomes, not all hotspots for carabids were also hotspots for All Coleoptera. The coincidence of hotspots for carabids, Coleoptera and other coleopteran taxon varied with taxon and biome, although there was overlap with all taxon in at least one biome. For example, although hotspots for carabids did not overlap with hotspots for curculionids and staphylinids in the Eastern Tiers or the Sanddunes, they did so in the Grasslands.

The proportional overlap of hotspots with hotspots is shown in Table 6.8. Carabid hotspots exhibited a positive non-random association with all other Coleoptera and the other four Coleopteran taxon although the proportional overlap varied (27% to 50%). The proportional overlap for carabids and All Coleoptera was 71%.

RESERVE SELECTION METHODS

The results of the three reserve selection methods in representing All Coleoptera and individual Coleopteran taxa among sites in three biomes are presented in Table 6.9.

The Hotspots approach

The Hotspots approach captured the fewest species (Table 6.9). This is unsurprising in regards to the Sanddunes, since only one site was defined as a hotspot for carabids. However, it did represent the same percentage (52%) of carabid and All Coleoptera species. In contrast, the five sites selected as carabid hotspots in the Eastern Tiers, although capturing 80% of carabid species, only ensured representation for 56% of All Coleopteran species. Hotspots for carabids in the Grasslands (5 sites) achieved proportional representation for Curculionidae and Scarabaeidae, but the proportional overlap for carabids and All Coleoptera was lower: 84% of carabid species were represented compared to 74% of all coleopteran species.

The Representative species richness (RSR) approach

Based on carabid SR from the hierarchical classification, the default group setting selected four sites in the Eastern Tiers, five sites in the Grassland biome, and three in the Sanddunes (Table 6.9). The Representative species richness (RSR) approach captured more species than the Hotspots approach and attained proportional representation for carabids and all Coleoptera in all three biomes, although the proportional overlap for individual Coleopteran taxa varied.

Complementarity: the greedy algorithm (GA) approach

The complementarity approach inevitably captured the largest number (84% to 96%) of all Coleopteran species and proportional overlap was high, although it still left some species unrepresented and required the greatest number of sites (Table 6.9). The percentage of sites representing all carabid species was the same for both the

Eastern Tiers and the Grasslands (39%) and ensured representation for a similar proportion of All Coleoptera and individual Coleopteran taxa. In the Sanddunes however, eight sites (80%) were needed to represent all Carabidae. However, this did ensure representation of 96% of All Coleopteran species.

To achieve the goal of representing all Coleopteran species at least once, 16 sites were required in both the Eastern Tiers and the Grasslands (89% and 70% of all sites in each biome respectively) and all sites were required in the Sanddunes. Almost all of the unrepresented species in each of the biomes could be classified as rare (occurred in < 20% of sites within a biome (Michaels 1999) and were represented by five or fewer individuals. Over half of all unrepresented species in each of the biomes occurred at only one site.

(a) Eastern Tiers

	Carabidae					
	Species counts			Rank order		
Taxon	r^2	P value	P	r^2	P value	P
All Coleoptera	0.24	0.04	<0.05	-0.26	0.285	NS
Coleoptera	0.41	0.004	<0.01	-0.41	0.09	NS
Curculionidae	0.33	0.01	<0.01	-0.39	0.11	NS
Scarabaeidae	0.24	0.04	<0.05	-0.43	0.08	NS
Staphylinidae	0.20	0.06	NS	-0.21	0.39	NS
Tenebrionidae	0.17	0.09	NS	-0.23	0.35	NS

(b) Grasslands

	Carabidae					
	Species counts			Rank order		
Taxon	r^2	P value	P	r^2	P value	P
All Coleoptera	0.65	0.000	<0.001	0.80	0.000	<0.001
Coleoptera	0.45	0.001	<0.001	0.60	0.002	<0.01
Curculionidae	0.23	0.02	<0.05	0.51	0.12	NS
Scarabaeidae	0.45	0.001	<0.001	0.69	0.000	<0.001
Staphylinidae	0.12	0.11	NS	0.34	0.11	NS
Tenebrionidae	0.14	0.09	NS	0.39	0.06	NS

(c) Sanddunes

	Carabidae					
	Species counts			Rank order		
Taxon	r^2	P value	P	r^2	P value	P
All Coleoptera	0.82	0.000	<0.001	0.82	0.001	<0.001
Coleoptera	0.67	0.003	<0.001	0.66	0.03	<0.05
Curculionidae	0.15	0.27	NS	-0.41	0.23	NS
Scarabaeidae	0.02	0.71	NS	0.18	0.61	NS
Staphylinidae	0.37	0.06	NS	0.06	0.86	NS
Tenebrionidae	0.18	0.23	NS	0.41	0.22	NS

Table 6.2. Pearson's correlation coefficients (r^2) for pairwise comparisons of total species counts and for pairwise comparisons of rank order correlation among sites between carabids and All Coleoptera, Coleoptera, Curculionidae, Scarabaeidae, Staphylinidae and Tenebrionidae for all three biomes.

TAXON	Site																	
CARABIDAE																		
rank	H1	H3	D1	AH1	H2	D3	AH2	AH3	B1	B3	T2	AG2	AG3	D2	T1	AG1	B2	T3
(SR)	10	10	8	8	8	6	5	4	4	4	4	3	3	3	3	2	2	2
ALL COLEOPTERA																		
rank	AG3	AH2	D1	AG2	B3	T2	AH3	B1	B2	D3	AG1	D2	T3	T1	AH1	H3	H1	H2
(SR)	52	52	51	50	50	49	48	48	46	39	38	34	31	30	29	25	24	20
COLEOPTERA																		
rank	AG3	AH2	AG2	B3	T2	AH3	B1	B2	D1	AG1	D3	D2	T3	T1	AH1	H3	H1	H2
(SR)	49	47	47	46	45	44	44	44	43	36	33	31	29	27	21	15	14	12
CURCULIONIDAE																		
rank	AH3	B1	AG2	B3	AG3	AH2	B2	T2	AG1	D1	D2	AH1	D3	T3	T1	H1	H2	H3
(SR)	13	12	11	10	9	9	9	9	8	8	8	6	6	5	4	3	3	2
SCARABAEIDAE																		
rank	T2	AG2	AG3	AH3	D1	AH2	B2	T1	AG1	B1	B3	T3	D3	D2	H3	AH1	H1	H2
(SR)	13	10	10	10	9	7	6	6	5	5	5	5	4	2	2	1	1	
STAPHYLINIDAE																		
rank	AH2	B3	B1	B2	D1	AG2	AG3	AH3	D3	AG1	T1	T2	H3	T3	AH1	D2	H1	H2
(S)	10	9	8	8	8	7	7	7	7	6	6	6	5	5	4	4	2	2
TENEBRIONIDAE																		
rank	D1	B1	B2	AG3	AH2	AH3	B3	T1	T2	AG2	AH1	D3	T3	D2	AG1	H1	H2	H3
(SR)	4	4	4	3	3	3	3	3	3	2	2	2	2	1	0	0	0	0

Table 6.3. Species counts by site and ranking of sites for each taxon based on species richness (SR) in the Eastern Tiers. See Chapter four for site descriptions.

TAXON	Site																						
CARABIDAE																							
rank	EFTE	CT	CRT	NTD	NGP	ND	CTP	NFP	CTT	CP	CTMP	EFT	EFD	0T	TP	NP	OFP	MRT	ORP	OTP	MID	TT	MIP
(S)	15	14	13	12	11	10	9	9	9	8	8	8	7	6	5	5	5	5	3	3	3	2	1
ALL COLEOPTERA																							
rank	EFTE	CRT	NTD	EFT	CP	NGP	CT	CTMP	ND	NP	CTT	EFD	CTP	OFP	ORP	OTP	TP	0T	NFP	MRT	TT	MIP	MID
(S)	75	60	55	52	51	50	50	45	44	41	41	41	40	40	38	38	37	31	30	25	23	17	13
COLEOPTERA																							
rank	EFTE	CRT	EFT	CP	NTD	NGP	CTMP	NP	CT	ORP	OTP	OFP	EFD	ND	TP	CTT	CTP	0T	NFP	TT	MRT	MIP	MID
(S)	60	47	44	43	43	39	37	36	36	35	35	35	34	34	32	32	31	25	21	21	20	16	10
CURCULIONIDAE																							
rank	CP	EFTE	CTMP	OFP	CRT	NTD	CT	CTT	ND	ORP	MIP	EFT	CTP	NGP	TP	NFP	NP	EFD	MID	TT	OTP	MRT	0T
(S)	11	10	10	8	8	8	7	7	7	6	6	6	5	5	5	5	5	5	5	4	3	2	2
SCARABAEIDAE																							
rank	EFTE	NGP	EFD	ND	NTD	EFT	0T	CP	CTP	NP	CT	CTMP	NFP	CRT	OTP	OFP	TP	TT	ORP	CTT	MRT	MIP	MID
(S)	14	14	13	13	12	9	9	8	8	8	8	7	7	7	6	6	5	5	4	4	3	2	1
STAPHYLINIDAE																							
rank	EFTE	NP	NTD	NGP	EFT	ORP	OTP	TP	OFP	ND	CRT	CP	CTP	CT	CTT	MRT	0T	EFD	CTMP	MIP	NFP	TT	MID
(S)	10	8	8	7	7	6	5	5	5	5	4	3	3	3	3	3	3	3	2	2	2	1	0
TENEBRIONIDAE																							
rank	EFTE	CTT	EFT	CTMP	CRT	EFD	ORP	OTP	CTP	TP	OFP	NTD	NGP	CT	0T	ND	NP	CP	MIP	NFP	MRT	TT	MID
(S)	5	5	5	4	4	4	3	3	3	3	3	3	2	2	2	2	2	1	1	1	1	1	1

Table 6 4. Species count for each site and site rank order for each group of taxa based on species richness (SR) in the Grasslands. See Chapter four for site descriptions.

TAXON	Site									
CARABIDAE										
rank	AB	FB	SC	WP	MB	WH	BB	TB	TE	EP
(SR)	15	10	10	10	9	9	8	8	7	6
ALL COLEOPTERA										
rank	AB	WH	FB	SC	WP	TB	MB	EP	BB	TE
(SR)	68	54	53	49	49	48	45	44	43	39
COLEOPTERA										
rank	AB	WH	FB	TB	WP	SC	EP	MB	BB	TE
(SR)	53	45	43	40	39	39	38	36	35	32
CURCULIONIDAE										
rank	WH	SC	TB	EP	BB	TE	FB	AB	WP	MB
(SR)	14	12	11	11	11	11	10	9	9	9
SCARABAEIDAE										
rank	MB	FB	TB	WP	BB	AB	SC	EP	WH	TE
(S)	8	7	7	6	6	5	5	5	4	2
STAPHYLINIDAE										
rank	AB	WP	TB	BB	TE	MB	EP	FB	WH	SC
(S)	8	7	3	3	3	2	2	1	1	1
TENEBRIONIDAE										
rank	FB	AB	WH	SC	BB	EP	TB	MB	TE	WP
(S)	10	8	8	8	6	5	4	4	4	2

Table 6.5. Species count for each site and site rank order for each group of taxa based on species richness (SR) in the Sanddunes. See Chapter Four for site descriptions.

TAXON	Site																	
CARABIDAE																		
rank	AG3	AG2	AG1	AH1	AH2	AH3	B3	B1	B2	D1	D3	D2	H3	H1	H2	T2	T1	T3
(SR)	3	3	2	8	5	4	4	4	2	8	6	3	10	10	8	4	3	2
ALL COLEOPTERA																		
rank	AG3	AG2	AG1	AH2	AH3	AH1	B3	B1	B2	D1	D3	D2	H3	H1	H2	T2	T3	T1
(SR)	52	50	38	52	48	29	50	48	46	51	39	34	25	24	20	49	31	30
COLEOPTERA																		
rank	AG3	AG2	AG1	AH2	AH3	AH1	B3	B1	B2	D1	D3	D2	H3	H1	H2	T2	T3	T1
(SR)	49	47	36	47	44	21	46	44	44	43	33	31	15	14	12	45	29	27
CURCULIONIDAE																		
rank	AG2	AG3	AG1	AH3	AH2	AH1	B1	B3	B2	D1	D2	D3	H1	H2	H3	T2	T3	T1
(SR)	11	9	8	13	9	6	12	10	9	8	8	6	3	3	2	9	5	4
SCARABAEIDAE																		
rank	AG3	AG2	AG1	AH3	AH2	AH1	B2	B1	B3	D1	D3	D2	H3	H1	H2	T2	T1	T3
(SR)	10	10	5	10	7	1	6	5	5	9	4	2	2	1		13	6	5
STAPHYLINIDAE																		
rank	AG3	AG2	AG1	AH2	AH3	AH1	B3	B1	B2	D1	D3	D2	H3	H1	H2	T2	T1	T3
(SR)	7	7	6	10	7	4	9	8	8	8	7	4	5	2	2	6	6	5
TENEBRIONIDAE																		
rank	AG3	AG2	AG1	AH2	AH3	AH1	B1	B2	B3	D1	D3	D2	H1	H2	H3	T2	T1	T3
(SR)	3	2	0	3	3	2	4	4	3	4	2	1	0	0	0	3	3	2
PCT		AG			AH			B			D			H			T	

Table 6.6. Species counts by site and rank order of sites based on species richness (SR) for each group of taxa within plant community type (PCT) in the Eastern Tiers. See Chapter Four for site descriptions and details of plant community type.

TAXON	Sites																				
CARABIDAE																					
rank	ORP	OTP	NGP	CTP	CP	CTMP	TP	MIP	NFP	NP	OFP	CT	CRT	CTT	EFTE	EFT	0T	MRT	TT	NTD	ND	EFD	MID
(SR)	3	3	11	9	8	8	5	1	9	5	5	14	13	9	15	8	6	5	2	12	10	7	3
ALL COLEOPTERA																					
rank	ORP	OTP	CP	NGP	CTMP	CTP	TP	MIP	NP	OFP	NFP	CRT	CT	CTT	EFTE	EFT	0T	MRT	TT	NTD	ND	EFD	MID
(SR)	38	38	51	50	45	40	37	17	41	40	30	60	50	41	75	52	31	25	23	55	44	41	13
COLEOPTERA																					
rank	ORP	OTP	CP	NGP	CTMP	TP	CTP	MIP	NP	OFP	NFP	CRT	CT	CTT	EFTE	EFT	0T	TT	MRT	NTD	EFD	ND	MID
(SR)	35	35	43	39	37	32	31	16	36	35	21	47	36	32	60	44	25	21	20	43	34	34	10
CURCULIONIDAE																					
rank	ORP	OTP	CP	CTMP	MIP	CTP	NGP	TP	OFP	NFP	NP	CRT	CT	CTT	EFTE	EFT	TT	MRT	0T	NTD	ND	EFD	MID
(SR)	6	3	11	10	6	5	5	5	8	5	5	8	7	7	10	6	4	2	2	8	7	5	5
SCARABAEIDAE																					
rank	OTP	ORP	NGP	CP	CTP	CTMP	TP	MIP	NP	OFP	NFP	CT	CRT	CTT	EFTE	EFT	0T	TT	MRT	ND	EFD	NTD	MID
(SR)	6	4	14	8	8	7	5	2	8	6	7	8	7	4	14	9	9	5	3	13	13	12	1
STAPHYLINIDAE																					
rank	ORP	OTP	NGP	TP	CP	CTP	CTMP	MIP	NP	OFP	NFP	CRT	CT	CTT	EFTE	EFT	0T	MRT	TT	NTD	ND	EFD	MID
(SR)	6	5	7	5	3	3	2	2	8	5	2	4	3	3	10	7	3	3	1	8	5	3	0
TENEBRIONIDAE																					
rank	ORP	OTP	CTMP	CTP	TP	NGP	CP	MIP	OFP	NP	NFP	CTT	CRT	CT	EFTE	EFT	0T	MRT	TT	EFD	NTD	ND	MID
(SR)	3	3	4	3	3	2	1	1	3	2	1	5	4	2	5	5	2	1	1	4	3	2	1
GCT	T1		T2						T3			T5			T6					T8			

Table 6.7. Species counts by site and rank order of sites based on species richness (SR) for each group of taxa within grassland community type (GCT) in the Grasslands. See Chapter Four for site descriptions and details of grassland community types.

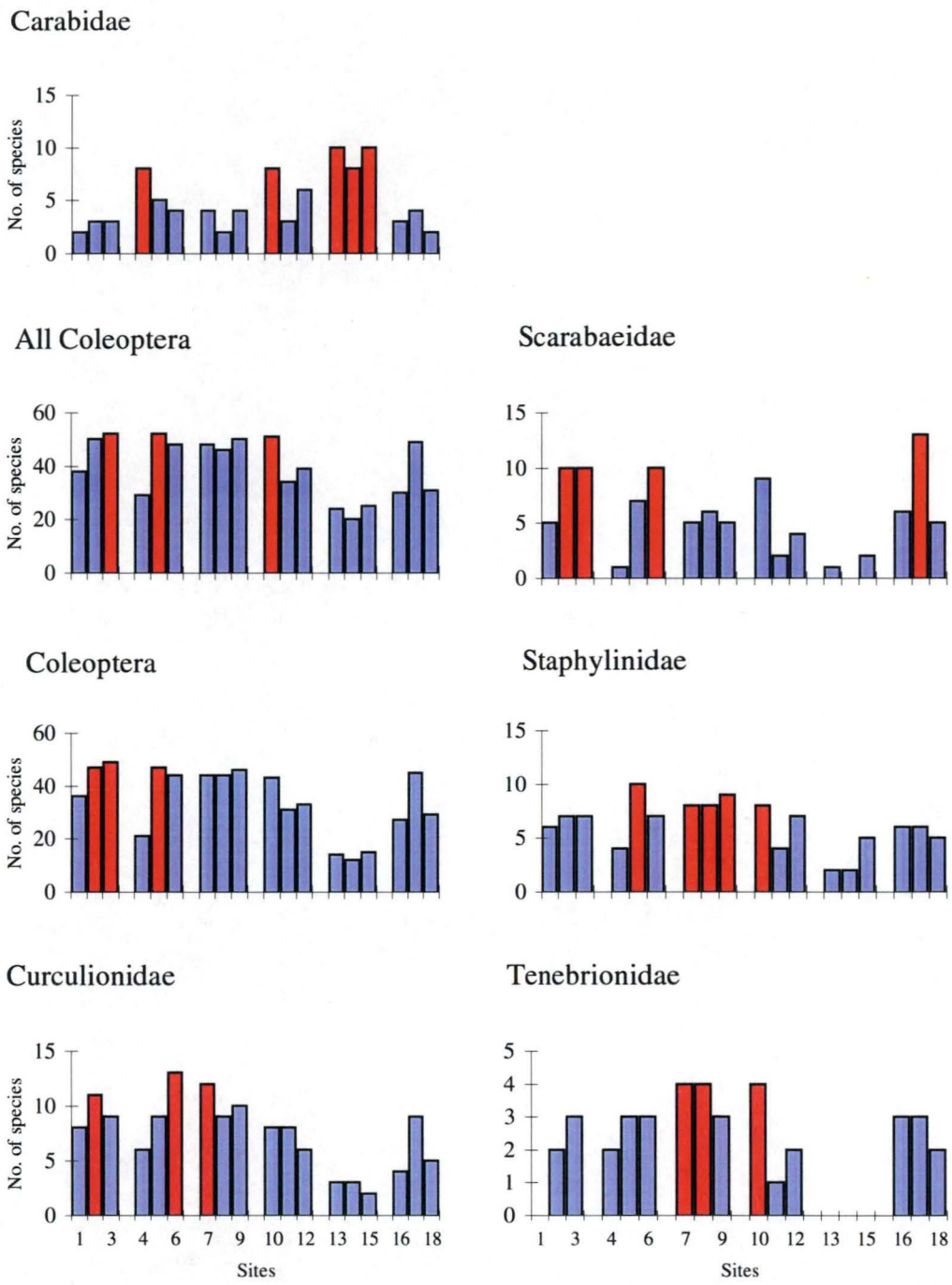


Figure 6.5. Coincidence of hotspots in the Eastern Tiers. Hotspots, shown for each group of taxa, are defined as mean SR plus 1 SD and indicated by red infill. See Chapter Four for the names and descriptions of sites.

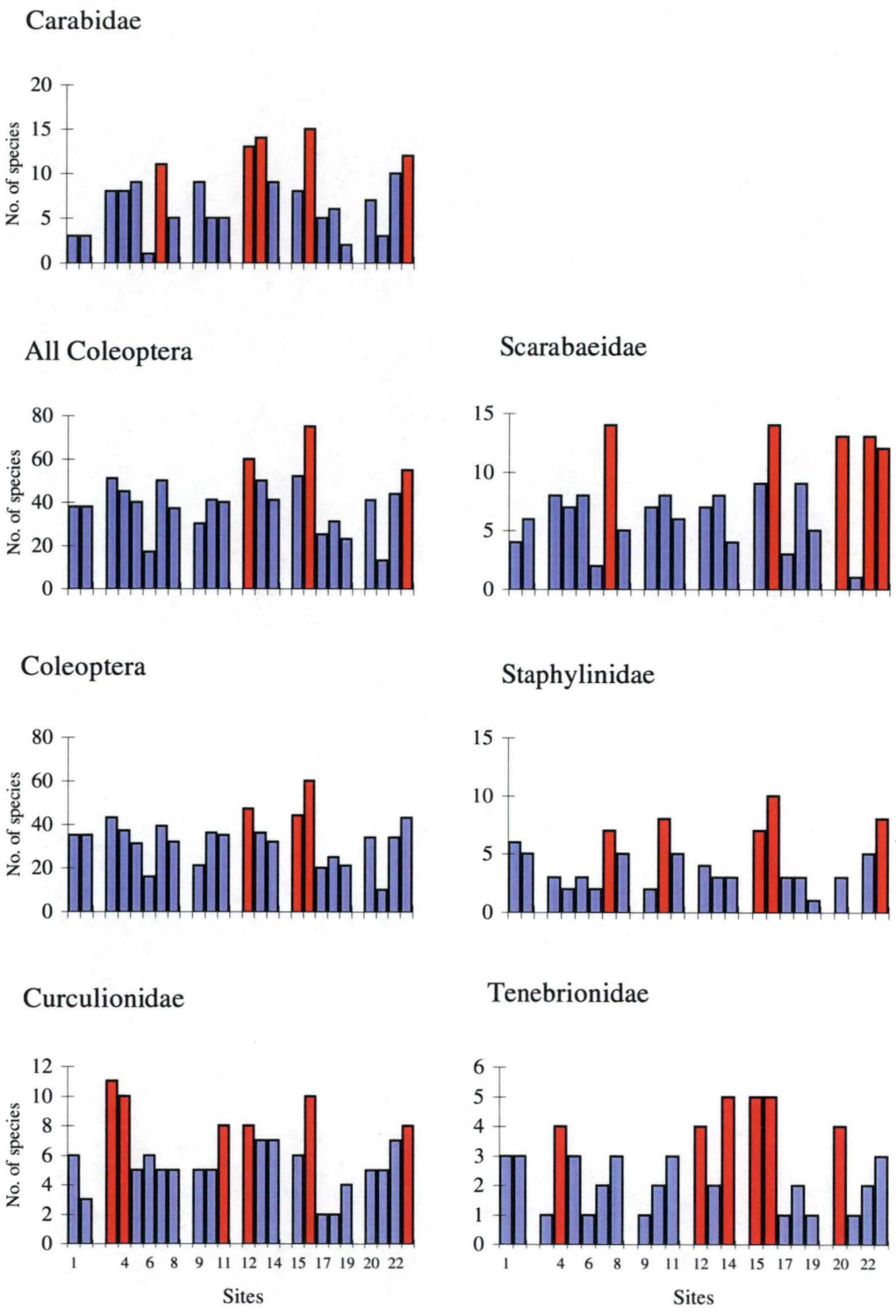


Figure 6.6. Coincidence of hotspots in the Grasslands. Hotspots, shown for each group of taxa, are defined as mean SR plus 1 SD and indicated by red infill. See Chapter Four for the names and descriptions of sites.

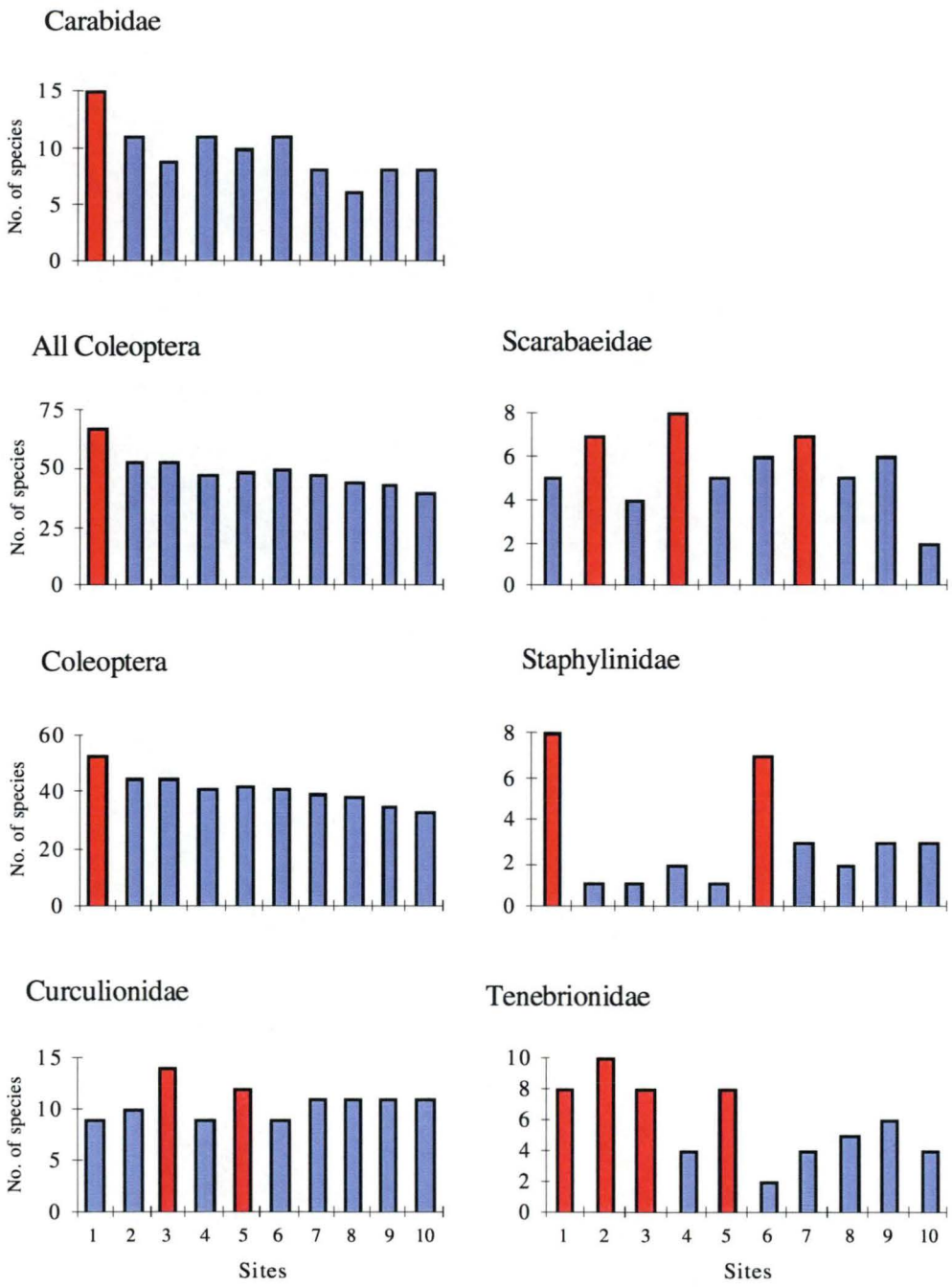


Figure 6.7 Coincidence of hotspots in the Sanddunes. Hotspots, shown for each group of taxa, are defined as mean SR plus 1 SD and indicated by red infill. See Chapter four for names and descriptions of sites.

TAXA	No. Hotspots	Carabidae	Probability
Carabidae	11	-----	
All Coleoptera	7	0.71 (n=7)	
Coleoptera	7	0.43 (n=7)	0.03
Curculionidae	11	0.27 (n=11)	0.05
Scarabaeidae	12	0.27 (n=11)	0.05
Staphylinidae	12	0.45 (n=11)	0.05
Tenebrionidae	13	0.50 (n=8)	0.05

Table 6.8. Proportional overlap of carabid hotspots with hotspots for All Coleoptera, Coleoptera and other Coleopteran taxa, n = maximum possible number of overlaps, calculated as the smaller of the pair of hotspot taxa adjusted for sites lacking records for either group. Probability of simultaneously selecting a hotspot for a given Coleopteran taxa and a carabid hotspot assuming no association based on percentage of sites defined as hotspots.

	Area-selection methods								
	Hotspots			Representative SR			Complementarity		
BIOME	ET	GL	SD	ET	GL	SD	ET	GL	SD
No sites	5	5	1	4	5	3	7	9	8
(%) selected	(28%)	(23%)	(10%)	(22%)	(23%)	(30%)	(39%)	(39%)	(80%)
Sites selected (rank order)	AH1 D1 H1 H2 H3	EFTE CT CRT NGP ND	AB	D1 AG2 H1 D2	EFTE CT CRT CTT CTMP	AB WP FB	H1 H3 AH1 AG2 T2 AH3 B1	EFTE CRT CT NGP NFP CTP TP NP MID	AB SC FB WP TE MB EP BB
Total species represented (%)									
Carabidae	80%	84%	52%	68%	84%	76%	100%	100%	100%
All Coleoptera	56%	74%	52%	67%	78%	75%	84%	89%	96%
Coleoptera	52%	72%	53%	67%	76%	75%	81%	86%	95%
Curculionidae	55%	86%	47%	73%	93%	63%	91%	86%	95%
Scarabaeidae	56%	80%	31%	72%	76%	69%	83%	88%	94%
Staphylinidae	64%	68%	100%	79%	63%	100%	86%	89%	100%
Tenebrionidae	50%	56%	42%	50%	78%	74%	70%	78%	95%
% spp Rare, ≤ 5 individuals and occurring 1 site unrepresented (UR)									
% Rare spp UR	60%	47%	69%	59%	44%	46%	32%	53%	10%
(%) UR spp	(68%)	(94%)	(58%)	(88%)	(100%)	(75%)	(96%)	(100%)	(100%)
% spp ≤ 5 inds UR	61%	46%	70%	59%	46%	46%	34%	20%	9%
(%) UR spp	(60%)	(69%)	(61%)	(78%)	(79%)	(78%)	(92%)	(80%)	(100%)
% spp 1 site UR	72%	62%	83%	61%	53%	59%	42%	45%	55%
(%) UR spp	(40%)	(67%)	(39%)	(45%)	(50%)	(53%)	(63%)	(49%)	(57%)

Table 6.9. Results of the three area-selection methods in representing Carabidae, All Coleoptera, Coleoptera and individual coleopteran taxa. Number and percentage (%) of sites selected, percentage (%) of species represented) and the percentage (%) of species, either Rare, represented by ≤ 5 individuals, or occurring at 1 site, unrepresented, under each selection approach, for sites in each of the three biomes.

DISCUSSION

Correlated species counts

The simplest test of the indicator status of a particular taxon/taxa is to examine species counts among sites for some pattern of covariation or correlation (Flather *et al.* 1997). The results demonstrate that although there was little or no significant covariation between species counts of carabids and individual Coleopteran taxa, there was significant positive correlation between carabid and All Coleoptera and between carabid and Coleoptera species counts in all three biomes, although the magnitude of the concordance varied with biome; the mean correlation among biomes was 0.57 and 0.51, (range 0.24 -0.82 and 0.41 - 0.67) respectively. Is this evidence of a significant pattern of covariation or correlation?

According to Pearson and Cassola (1992) and Flather *et al.* (1997), yes. Pearson and Cassola (1992) considered that a significant ($P = 0.01$) mean correlation of 0.54, range (0.36 - 0.73) between tiger beetle counts and bird counts at a continental scale was evidence that tiger beetles qualified as biodiversity indicators. While Flather *et al.* (1997) consider significant correlations of 0.5 or less show only weak concordance; correlations of 0.79 stronger, they still consider both as evidence of covariation. Indeed, any significant correlations >0 ($P<0.05$) can be considered evidence of some covariation.

The primary reason for interest in correlation between species counts is the need to identify priority sites of high diversity, with those areas exhibiting high species numbers across many taxa given the highest priority (Vane-Wright *et al.* 1991). A significant correlation between the patterns of species richness exhibited by an 'indicator group' and the more general patterns such as the comparison and ranking of sites or regions in terms of their species richness is considered sufficient for a group of this type to serve the purposes of biodiversity 'assessment' (Hammond 1994).

Cranston and Trueman (1997) advocate that biodiversity predictors (indicators) should predict the diversity of other taxa and that evidence of this would be congruence between site rankings. Although they found no congruence in site rankings of five Tasmanian sites based on species counts for eleven different

terrestrial arthropod taxa they did find that the site rank order for plant species exactly matched that for ants and concluded that this was evidence that ant species diversity and plant species diversity was well related.

Although the instances where site rankings on species richness per site agreed with site rankings based on carabid species richness varied with Coleopteran taxon and biome, there were many points of agreement, with the proportional overlap of agreed site rankings (calculated as the number of concurring site rankings over the number of possible site rankings) ranging from 17% to 58%. The proportional overlap of agreed site rankings between carabids and All Coleoptera and between carabids and Coleoptera was 42% and 58% respectively. These results would seem to indicate a definite relationship exists between the patterns of species richness exhibited by carabids and other Coleoptera and that carabid species diversity and overall Coleopteran species diversity are relatively strongly related.

Agreement between the proposed biodiversity indicator and other taxon on the question "which site is most diverse" is also considered evidence that a taxon can be used as predictor (Cranston and Trueman 1997). Although there was limited agreement on which site was most diverse on species counts of carabids and individual Coleopteran taxa in all three biomes, the most diverse site for All Coleoptera (i.e. the site with the highest number of all Coleoptera species) was predicted by carabid species richness in the Grasslands and in the Sanddunes, although not in the Eastern Tiers, in other words, it was predicted in two out of the three biomes studied. Carabid diversity also predicted the most diverse site in five out of the six plant community types in the Eastern Tiers and three out of the six grassland community types in the Grasslands. In other words, carabid species richness predicted the site with the highest Coleoptera species richness within biomes and within vegetation community types in the majority (67%) of instances.

It has been suggested that correlations between the patterns of species richness exhibited by an 'indicator group' and other taxa are least likely where substantial differences in lifeways and significant features such as body size are involved (Hammond 1994). Conversely, it could be expected that correlations should be most likely where such differences are reduced, at the order level for instance. My results give strong support to this hypothesis. This assumption is also supported by the results of Beccaloni and Gaston (1994), who found that the diversity of the Lepidopteran subfamily Ithomiinae correlated well with total butterfly diversity. Similarly, while Oliver and Beattie (1996) found no significant positive correlations between ant, beetle, and spider species richness, they did find that richness within

the family Carabidae was significantly correlated with richness within all other beetle families.

Hotspots

Flather *et al.* (1997) suggest that an appropriate test of an indicator taxon is examining the data for evidence of geographic coincidence between hotspots of an indicator taxa and other taxa. It is not crucial that the full range of species distributions covary, but rather that there is some degree of spatial concordance among areas that are particularly rich in species (i.e. are hotspots) for different taxonomic groups (Flather *et al.* 1997). But what degree of concordance is crucial?

The probability of randomly selecting a hotspot (defined here as mean SR plus one SD) from two distinct, normally distributed, populations is 3%. Conclusions drawn about values between 3% and 100% are qualitative and the point at which concordance makes a taxon a good indicator has yet to be defined. Examining hotspot overlap for five taxa, Prendergast *et al.* (1993) found that there was some positive non-random concordance between most pairs of taxa. The exception being liverworts, which had no hotspots in common with either butterflies or dragonflies. Prendergast *et al.* (1993) called an overlap of 34% between hotspots for butterflies and dragonflies weak. The overlap of 60% between Butterfly and Aquatic plant hotspots (Table 2 (a) p 337) was ignored, but would they have considered it strong had they noticed it?

The proportional overlap between carabid and All Coleoptera hotspots was 71%. It is expected that the effects of auto-correlation will lead to higher proportion of overlap between carabids and All Coleoptera than that between carabids and other Coleoptera (43%) or between carabids and individual Coleopteran taxa (27% to 50%). Still, seven out of ten carabid hotspots were also hotspots for All Coleoptera. I suggest that this concordance is strong enough to accept carabids as a biodiversity indicator for ground-dwelling Coleoptera in general.

Complete concordance between species counts for different taxa is unlikely to occur, particularly at the local (site) scale for a number of reasons. Species richness is related to complex environmental and ecological variables (O'Connor *et al.* 1990). The factors influencing the distribution of different taxa can be expected to vary, since different species have different ecological preferences. Indeed, Prendergast *et al.* (1993) suggested that one basis for the incongruence of the

different indicator groups in their study was that they tended to favour different environments, and exhibited different preferences for temperature and moisture.

Species diversity at the local scale will be influenced by both the species diversity of surrounding area and the species diversity at the regional level. The regional species pool will contain a mix of species with varying habitat preferences, dispersal abilities and migratory habits. Migrating and transient species from neighbouring faunas may increase the local fauna. For example, an increase in the invertebrate diversity of heathland patches was attributed to migration from surrounding vegetation (Webb and Hopkins 1984; Webb 1989), and Bauer (1989) found an active interchange of beetles across a limestone, grassland-blanket peat interface. In this study, it is possible that some beetles may have been temporary migrants adding to the species count in much the same way as the annual migration of some bird species leads to an increase in species richness at particular times of the year in certain places (Michaels and Mendel 1998).

The proportion of vagrants, transients or migrants can be high. Almost half the arthropod species encountered in a study of alfalfa were classified as vagrants (Pimentel and Wheeler 1973); and a fifth of the beetle species from oak trees sampled by fogging were classed as tourists (Gaston *et al.* 1993). In this study, 22% to 51% of species could be classified as rare, based on distribution (that is, they occurred in less than 20% of sites), around 40% of all species were represented by fewer than five individuals and a fifth to a quarter of all species recorded occurred at just one site (Table 6.9). However, without detailed knowledge of the distribution of Coleopteran species, it is impossible to determine the difference between genuinely rare species and vagrants, transients or migrants.

The relative contribution of vagrants to species richness will also depend on the taxon, the habitat, the size of the study area and the duration of the study, with the proportion of species which are vagrants tending to increase with the mobility of the taxon and the duration of the study (Desender 1996; Gaston 1996). Many of the Coleopteran species in this study are winged and thus comparatively mobile. The proportions of species in different taxa may also change with the scale of observation (Weaver 1995). Weaver (1995) observed that some taxa added species at the same rate as sample area increased; some decreased in species richness with increased scale; while others increased with scale (i.e. new samples added new species). The rate of increase was low and variable for all but Coleoptera, which rapidly added new species going from 4% of the species in a sample to 22% of the total. She suggested one explanation as being that different vagrant species of

Coleoptera contaminated different litter samples more often than vagrants of other groups and that many species demonstrated patchy distributions. In this study, sampling effort was consistent across sites within biomes, and all sites were sampled for a minimum of one year so it can be assumed that sampling accounted for species from a similar sized area and therefore accumulated species at a similar rate and that species uniform in space, but patchy in time would still have been accounted for. Nevertheless, the local species pool will be a composite of residents, vagrants, transients and migrants and this needs to be kept in mind when considering species counts (Desender 1996; Gaston 1996).

Reserve selection

The non-overlap of sets of hotspots for different taxonomic groups demonstrates the difficulties in using indicators to choose a limited number of reserve areas that are to protect many different organisms (Prendergast *et al.* 1993). However, although sets of individually species rich areas may not be predictive of sets of hotspots for other groups, a set of areas, properly selected could be generally species rich as a set (Faith and Walker 1996b). For example, Prendergast *et al.* (1993) observed that if every hotspot for just one of the taxon in their investigation were protected, more than half the species in the targeted taxon and of species in every other group would also occur in the protected areas. Indeed, the set of bird hotspots contained more butterfly species than did the set of butterfly hotspots. In this study, the set of carabid hotspots also contained more than half of all coleopteran species. However, the set of carabid hotspots did not ensure proportional representation for other coleopteran taxa. This highlights one of the limitations of the hotspot approach, that collections of individually species rich sites will not necessarily conserve the range of diversity, or even all of the target species (Kirkpatrick 1983; Faith and Walker 1996b; Flather *et al.* 1997).

Results of the three reserve selection methods show that although the set of carabid hotspots did not ensure proportional representation for all Coleoptera, the Representative SR approach and the complementarity approach did. However, although ensuring proportional representation, the Representative SR approach did not contain all coleopteran species. Similarly, Margules *et al.* 1988) found that the most species rich wetlands from each of nine wetland type, while selected to ensure full representation of major wetland habitat types in northern New South Wales, included only 67% of all native wetland plant species.

However, a set of representative areas in which one taxon is completely represented may represent the diversity of other taxa (Csuti *et al.* 1997). In a test where the initial selection of protected areas was based on the representativeness criteria, Game and Peterken 1984) found that if all sites in central England containing rare plants were selected, 99% of all other plants were conserved. Since the Representative SR approach based on carabids gave proportional representation to all Coleopteran species, it is likely that a set of representative areas in which carabids are completely represented will substantially represent the diversity of other ground dwelling Coleopteran taxa. This expectation is supported by the results of the complementarity approach, where representation for all carabids gave protection to over 90% of all Coleopteran species.

Complete representation for all Coleoptera species would have required a much larger number of number of sites. For example, in the Sanddunes, all sites would be needed. Similarly, Stokland (1997) in a study of representativeness for bird and beetle conservation in Norwegian Boreal forests, found that all beetle sample sites had unique species, and consequently all were necessary to include all (509) species, although 90% of the species were encountered in the first 10 (of 17) sites using iterative selection.

Beyond a certain number of sites, adding additional sites yields an increase of only one or two species in all richness based algorithms (Csuti *et al.* 1997). Moreover, the number of sites needed to represent all taxon can vary greatly (Ryti 1992). Mobile, habitat generalists with a wide geographic distribution will need the fewest sites to be fully represented, while more sedentary habitat specialists with restricted distribution will need a greater proportion of sites (Csuti *et al.* 1997). In this study, 99% of unrepresented species in the Complementarity approach had a restricted distribution, that is, they occurred in less than 20% of all sites, and over half occurred at only one site.

The number of different organisms to be represented will also influence the proportion of sites required to represent all species. For example, when woodlands were selected iteratively in Norway, only 20% of the total area was need to incorporate all bird species compared to 75% to include all plant species (Saetersdal *et al.* 1993). Similarly, to include all 8 bird species found in the San Diego canyons in San Diego County, California, required only 2 canyons, 3.2% of the total area, while to include all 87 native plants required 10 canyons, 66% of the total area (Ryti 1992). In Natal, South Africa, 65 carnivore and ungulate species could be represented in nine out of 166 quarter degree grid squares chosen by an iterative

richness algorithm, yet representation for the 570 bird species required 27 grid squares, and the 6111 plant species required 140 of the 160 sites for total representation (Csuti *et al.* 1997). While in the Norwegian boreal forests, Stokland (1997) found that all 56 bird species were encountered in 14 out of 40 sites, but the 509 beetle species required 17 out of 17 sites.

CONCLUSION

Does this study offer support for using carabids as biodiversity indicators to reflect overall Coleopteran (defined in the context of this thesis as ground-dwelling Coleoptera) diversity patterns? In other words, did diversity measures of carabids predict the diversity of other Coleopteran taxa or the diversity of all Coleoptera in general? Results for the two test measures of indicators (i.e. correlated species counts and coincidence of species "hotspots") demonstrated that although there was limited covariation between species counts of carabids and individual Coleopteran taxa, occurring in some instances but not others, there was a significant positive pattern of covariation of species counts for carabids and all Coleoptera in general. Moreover, carabid species richness predicted the site with the highest Coleoptera species richness within biomes and within vegetation community types in the majority of instances.

If the goal is to identify the minimum number of areas that represents the range of variation and has as a set high biodiversity, a test of the biodiversity indicator concept is: will using carabid data identify such a set? An effective biodiversity indicator taxon in the context of representativeness would be one for which a set of representative areas will ensure proportional representation for other Coleoptera. The Representative species richness (RSR) approach based on carabids gave proportional representation to all coleopteran species. An effective biodiversity indicator taxon in the context of complementarity would be one for which a set of areas species rich for the indicator group will be species rich in general (Faith and Walker 1996b). The results of the complementarity approach demonstrated that a set of areas species rich for carabids were also species rich in general for all Coleoptera, since representation for all carabids gave protection to 90% of all Coleopteran species. It is therefore likely, that a set of representative areas in which carabids are completely represented will substantially represent the diversity of other ground-dwelling Coleoptera.

The results of this thesis support the hypothesis that the family of ground beetles (Carabidae) is an appropriate biodiversity indicator taxon for identifying and predicting the biodiversity patterns of all ground dwelling Coleoptera in most but not all instances. It also supports the hypothesis that reserve selection based on carabids will provide substantial protection for other ground dwelling Coleoptera.

Chapter Seven

Assessing the indicator potential of Carabids for monitoring: Carabids as Ecological indicators

Monitoring key indicators such as components of biological diversity has been identified as essential if we are to assess accomplishment of the goal of sustainable management (Kremen *et al.* 1993; Vora 1997; Bertollo 1998). If we are to successfully change and modify our actions, we need ecological indicators to measure the impacts of our management strategies and point out how well we are doing at maintaining biological diversity (Vora 1997), we need to monitor to assess the success or otherwise of our management practices (Jackson and Fox 1996), and we need to be able to predict the potential impacts of human activities on biological diversity across a hierarchy of spatial and temporal scales (Hansen *et al.* 1993; Vora 1997; White *et al.* 1997).

Forests are a major part of Australia's biodiversity, covering nearly 20 percent of Australia's land mass (Hill 1998). Ecologically Sustainable Management (ESM) forms part of the National Strategy for Ecologically Sustainable Development (Australia 1992). The Australian government through Regional Forest Agreements has stated that it will ensure that forests are harvested according to principles of ecologically sustainable management (Hill 1998). Long term sustainability of biological diversity is dependant on the maintenance of viable fauna populations within managed landscapes (Noss and Harris 1986; Hansen *et al.* 1991; White *et al.* 1997; Vora 1997). Therefore, successful maintenance of populations not just habitats is the ultimate measure of success in ecologically sustainable forestry (Thompson and Welsh 1993; McLaren *et al.* 1998).

Commercial forestry practices constitute a major ecological impact on forest ecosystems (Kirkpatrick and Bowman 1982; Hansen *et al.* 1991; Norton 1996). Currently, in Tasmania, clear-felling is the dominant silvicultural practice for both sawlog and pulp production. Clear-felling (removal of the whole tree layer from coupes) and slash-burning initially reduces structural complexity and the consequent replacement of ecologically complex forest with single-aged stands, without dead or

overmature trees can have profound effects and may reduce the diversity and/or alter the composition of forest invertebrates (Niemelä *et al.* 1993a; Niemelä *et al.* 1993b; Michaels and McQuillan 1995; Spence *et al.* 1996; Niemelä 1997; Michaels and Bornemissza 1999). Moreover, due to the variety of habitat types in old-growth forest, clear-felling may have varying effects on forest invertebrates species composition and diversity (Heliövaara and Väisänen 1984; Haila *et al.* 1994).

Studies on the general response of forest invertebrate communities to logging, or on their recovery with regeneration are scarce. Many such studies have focussed on carabids (e.g. Lenski 1982; Holliday 1992; Niemelä *et al.* 1993a; Niemelä *et al.* 1993b; Michaels and McQuillan 1995; Spence *et al.* 1996; Attlegrim *et al.* 1997; Niemelä 1997; Taylor *et al.* 1999), but there is a growing body of work on forest management practices and saproxylic beetles in the temperate areas of the Northern Hemisphere (see Martikainen *et al.* 1996; Økland *et al.* 1996; Kaila *et al.* 1997) and in Tasmania (Meggs 1996b; Michaels and Bornemissza 1999).

Noss and Cooperrider (1994) suggested that the selection of indicators may be the most vital component of establishing a monitoring program for adaptive forest management. In Canada, forest managers plan to use (fauna) indicators to monitor and assess the results of forest management as part of a legal requirement to demonstrate that forest harvesting is conducted in a manner not detrimental to the sustainability of forest ecosystems (McLaren *et al.* 1998). The proposal is to use indicators to assess changes within habitats, ecosystems and landscapes (Mclaren *et al.* 1998). The objective is to test the general hypothesis that forest management has no effect on species richness and species abundance or the distribution of species in time and space (Mclaren *et al.* 1998). The assumption is that the response to forest management exhibited by the selected indicator species reflects that of other unmonitored taxa or at least a subset of other taxa, i.e. that they are ecological indicators, yet this assumption has still to be tested.

AIMS

The aims of this chapter are:

- to determine the response of carabids, ground-dwelling Coleoptera in general, and other coleopteran families to commercial forestry harvest and regeneration practices in eucalypt forests in Tasmania (clearfelling followed by burning and regeneration).

- to compare the responses of the carabid fauna to the responses of ground-dwelling Coleoptera in general, and to the responses of other coleopteran families to assess if the responses of carabids to this environmental disturbance are representative of the response of ground-dwelling Coleoptera in general, and/or of other coleopteran families.

- to assess the utility of carabids as ecological indicators for other selected ground-dwelling Coleoptera

METHODS

STUDY SITES

Samples taken from a chronosequence of sites regenerating after a single cycle of clearfell and burning in the tall wet sclerophyll forests of southern Tasmania in the Picton State Forest, and the dry sclerophyll forests of eastern Tasmania in the Weilangta State Forest, are used to study the effects of logging on the species richness, abundance and composition of the carabid and other coleopteran assemblages in each forest type. In the text, the groups of sites from each forest will be referred to as the Picton and Weilangta respectively.

Both forests have been subjected to selective logging over the last century and to the current forestry practice of clearfell and slash-burn since the 1970's. Clear-felled coupes are typically around 50-80 ha and are dispersed through the landscape. Wet forest coupes yield around 300 tonnes/ha of pulpwood and sawlogs with up to 200 tonnes/ha left on site as residual slash and stumps prior to burning (J. Traill, pers. comm. 1994). Dry forest yield is lower, at around 200-240 tonnes/ha with 30-60 tonnes of residual slash and stumps (B. Warren, pers. comm. 1998).

SAMPLING

Sampling of the Coleopteran fauna was undertaken in old-growth forest and a chronosequence of coupes regenerating subsequent to clear-fell and burning, duplicated on age class (time since cutting), in both forest types. Regeneration age classes were based on time since clearfelling. There were three regeneration age classes in the wet sclerophyll forest: Early (1- 4 years), Intermediate (9 years) and Late (25 years) plus two old-growth control (Old-growth) sites. In the dry sclerophyll forest the three regeneration age classes were Early (1 years), Intermediate (9 years) and Late (20 years), plus 6 old-growth forest sites. See Chapter Four for site descriptions and Chapter five for sampling protocol. The terms *regenerating* or *regrowth* will be used to refer to secondary forest sites that are developing after the previous forest was clear-cut.

DATA ANALYSIS

Data analyses were applied to carabids, all Coleoptera sampled, including carabids (referred to as All Coleoptera), Coleoptera excluding carabids (referred to hereafter

as Coleoptera) and four coleopteran families that occurred in both forests, namely: Curculionidae, Lucanidae, Staphylinidae and Tenebrionidae. The proportions of species in each family in each forest are shown in Figure 7.1.

Within site (alpha) and between-site (beta) diversity (Whittaker 1977) was used to examine assemblage-level variation for Carabidae, All Coleoptera, Coleoptera, Curculionidae, Lucanidae, Staphylinidae and Tenebrionidae, in and between the samples (grouped age classes).

Alpha diversity

Alpha diversity was measured as species richness (SR), the number of species present at a site. The distribution patterns of individual species of each taxon in each forest were examined using presence and abundance data from each site.

Beta-diversity

Beta diversity was measured as the mean species richness (SR) of all sites within an age class. For each coleopteran taxon, in each forest, means were computed for species richness and abundance within age class and comparisons made between age classes.

The Morisita Horn Index of Similarity

The Morisita-Horn Index of similarity (after log transformation (LN(x+1)) (Wolda 1981) was used to investigate variations in the species composition of assemblages across the successional gradient. The values of the Index range from 0, when the two communities have no shared species, to a maximum of 1 when the species composition and relative abundances are identical in both communities.

The expected maxima of many similarity indices are strongly dependent on sample size and the effect increases with faunal diversity. While the Morisita index is independent of sample size and diversity (except possibly for very low sample sizes Wolda 1981), and this major advantage suggests that it should be used in preference to other indices (Wolda 1981), where logarithmic transformation of the data is required, for example, to improve normality and homogeneity of variance in the data, as in this thesis, the use of the Morisita-Horn index is more appropriate (Wolda 1981).

An analytical problem particularly pertinent to the analysis of invertebrate data is the problem of species represented by few individuals, often (perhaps inappropriately) labelled as "rare" species (Oliver *et al.* 1999). Sometimes, these species are omitted from analyses, however, it may be these species that are most sensitive to environmental perturbation (Faith and Norris 1989; Oliver *et al.* 1999). Therefore, all species were included in these analyses.

The INDVAL index

Characteristic or 'indicator species" and indicator species assemblages for successional stages and Old-growth sites were identified using the INDVAL index developed by Dufrene and Legendre (1997) to identify symmetrical indicators. Symmetrical indicators (*sensu* Dufrene and Legendre 1997), are defined as the most characteristic species of each group, found mostly in a single group of the typology and present in the majority of the sites belonging to that group, their presence contributes to the habitat specificity and one can therefore, predict their presence in all sites of that group (habitat type).

A species not present at the majority of sites in one habitat group, but present only in that group (i.e. rare and specialist species) is an asymmetrical indicator (*sensu* Dufrene and Legendre 1997), although its presence cannot be predicted in all sites of that group (habitat type), its presence contributes to the habitat specificity.

The INDVAL index for each species *i* in each site group *j*, is calculated as follows:

$$INDVAL_{ij} = A_{ij} \times B_{ij} \times 100$$

Where INDVAL is the Indicator value of species *i* in site cluster *j*,

Aij = N individuals *ij*/ N individual *si* and

Bij = N sites *ij*/N sites

and the final multiplication by 100 produces percentages. Quantities A and B are combined by multiplication because they represent independent information about the species distribution (Dufrene and Legendre 1997).

In the formula for Aij, which is a measure of specificity, Aij, is the mean abundance of species *i* in the sites of group *j* compared to all groups in the study. N

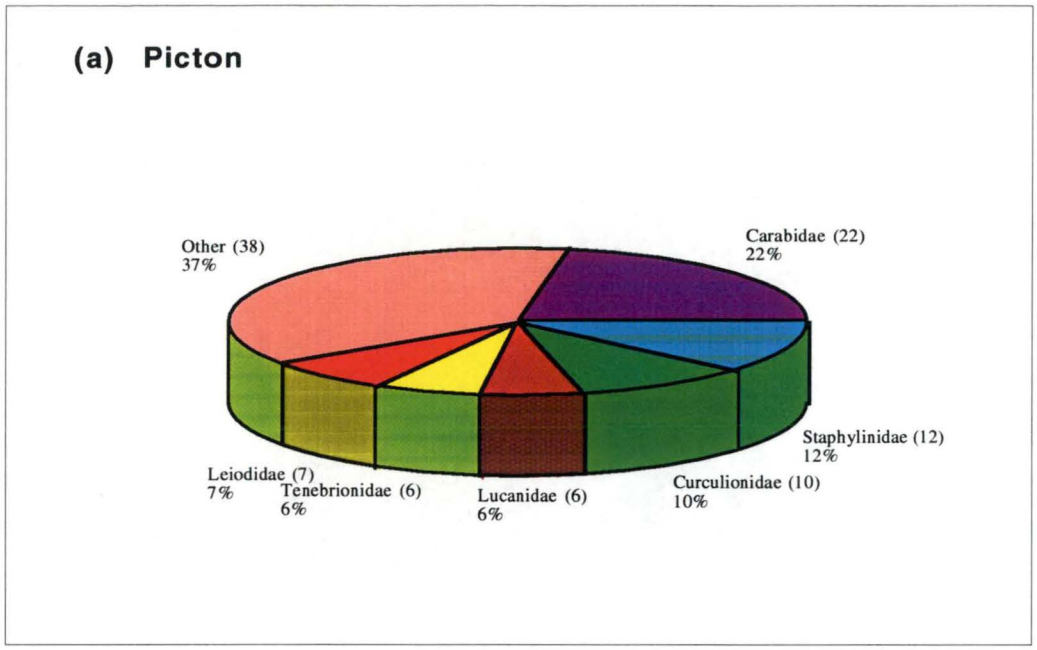
individuals *ij* is the mean number of individuals of species *i* across sites of group *j* and N individual *si* is the sum of the mean numbers of individuals of species *i* over all groups. Aij is maximum when species *i* is only present in cluster *j*. The mean number of individuals in each group is used instead of summing the individuals to remove any effect of the number of sites in the various groups and of differences in abundance in sites belonging to the same group.

In the formula for B*ij*, which is a measure of fidelity, B*ij*, is the relative frequency of occurrence of species *i* in the sites of group *j*. N sites *ij* is the number of sites in cluster *j* where species *i* is present and N sites *j* is the total number of sites in that cluster. B*ij* is maximum when species I is present in all objects of cluster *j*.

A characteristic species is one with an Indicator value of 25% or more. This supposes that a characteristic species is present in at least 50% of one site group and that its relative abundance in that group reaches at least 50% (Dufrene and Legendre 1997). A symmetrical indicator is one with an Indicator value of 55% or more (Dufrene and Legendre 1997). All individuals except those represented by a single individual were included in the analysis. The INDVAL index for species in the Intermediate and Late successional stages and Old-growth was computed excluding the Early successional stage, since the presence of old-growth or late successional species in recently clearfelled sites may indicate survival post logging rather than habitat preference (Michaels and Bornemissza 1999).

UPGMA analysis

The similarity of the species composition at the different areas was examined using the agglomerative clustering technique "Flexible UPGMA" from the PATN (Pattern Analysis Package) software package (Belbin 1995b). The input is a similarity matrix derived from the species data based on the rank order of all dissimilarities and the output is a dendrogram displaying the results of clusters of associated objects (sites). The Bray-Curtis similarity co-efficient was used to construct the matrix. The abundance of each species caught in each site was transformed to a percentage of the total abundance of all species of the taxon to which it belonged, or group of taxon (i.e. Coleoptera or All Coleoptera) in that site thus converting the data to a proportion between 0 and 1. The default group setting, Group Definition (GDEF) in PATN (Belbin 1995b) was used to determine the number of defined groups or clusters. A Mantel test was used to assess the degree of resemblance between matrices for pairwise comparisons between carabids and All Coleoptera, Coleoptera, Curculionidae, Scarabaeidae, Staphylinidae and Tenebrionidae in each forest.



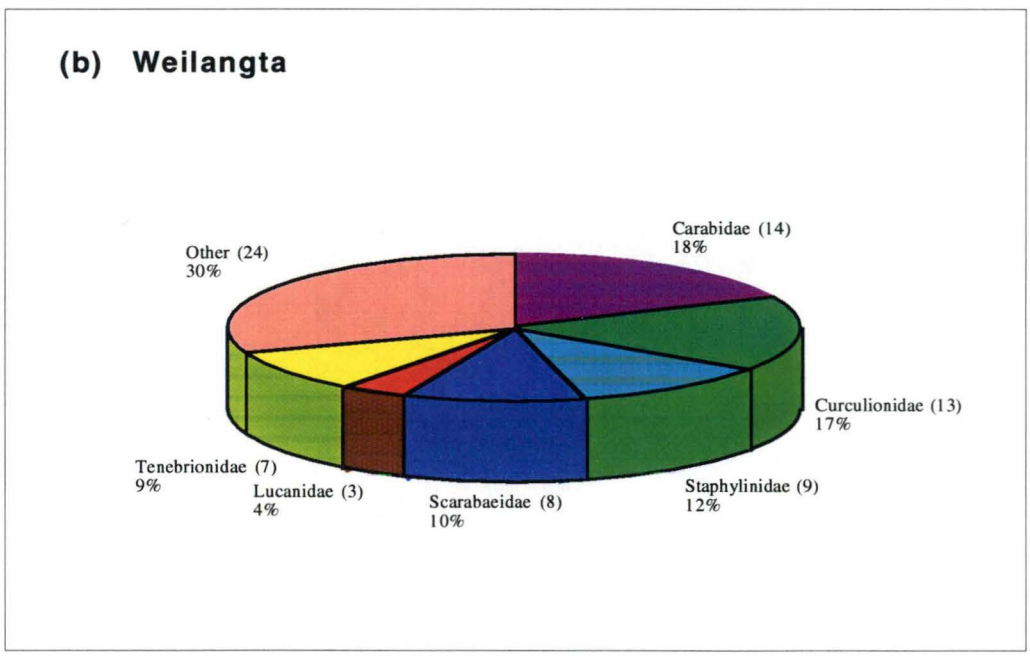


Figure 7.1. The number of species in each Coleopteran family and the contribution (%) made by each family to the total catch in (a) the Picton forest and (b) Weilangta forest.

RESULTS

VARIATION ACROSS THE SUCCESSIONAL GRADIENT

Diversity

Picton

A total of 3466 individuals and 99 species of Coleoptera, of which 1184 individuals and 22 species were carabids were recovered from the traps. Forty species, including 10 carabid species, were represented by 5 or fewer individuals and were therefore considered rare (Holliday 1992; Michaels and McQuillan 1995).

The species richness of Picton Coleoptera showed two general patterns of response to logging, which were taxon specific (Fig. 7.2.):

(1) the highest species richness in the Early stage of regeneration: this was shown by Carabidae, All Coleoptera and Lucanidae, or

(2) the highest species richness in Late regeneration sites: shown by Coleoptera, Curculionidae, Staphylinidae, and Tenebrionidae .

The abundance of Picton Coleoptera varied in response to forest management, and was taxon specific (Fig. 7.3.):

(1) lucanids were most abundant in the early regrowth sites,

(2) carabids were most abundant in the Intermediate regeneration sites: carabids,

(3) All Coleoptera, Coleoptera, Staphylinidae and Tenebrionidae were most abundant in Late regeneration sites, and

(4) curculionids were more abundant in Old-growth sites.

Weilangta

A total of 1658 individuals and 78 species of Coleoptera, of which 836 individuals and 14 species were carabids, were recovered from the traps in Weilangta State

Forest. Thirty-four species, including 4 carabid species, were represented by 5 or fewer individuals and were therefore considered rare.

The species richness of Weilangta Coleoptera showed two general patterns of response to logging, similar to that in the Picton, but again taxon specific (Fig. 7.4.):

(1) the highest species richness in the Early stage of regeneration: this was exhibited by carabids, All Coleoptera, Coleoptera, curculionids and lucanids; or

(2) the highest species richness in Late regeneration sites: shown by staphylinids

Tenebrionids were equally species rich in Early and Late regrowth.

Similarly, the abundance of Weilangta Coleoptera showed two general patterns of response (Fig. 7.5.):

(1) carabids, All Coleoptera, Coleoptera and Lucanidae were most abundant in the Early regeneration sites; while

(2) Curculionidae, Staphylinidae and Tenebrionidae were most abundant in Late regeneration sites.

Species composition

Picton

Between Age classes

While the Morisita-Horn index values for community composition between age classes for Coleopteran taxon in the Picton showed some variation, most markedly for carabids (range 0.33 to 0.87) there were several obvious patterns (Table 7.1).

(1) For all taxa, community similarity between Early and Intermediate and Early and Old-growth was greater than between Early and Late, (showing evidence of survival and short term persistence followed by absence from late of many species).

(2) For all taxa, except Carabidae, community similarity was greater between Intermediate and Old-growth sites than between Intermediate and Late.

(3) For all taxa, except Staphylinidae, there was greater similarity in species composition between Intermediate and Old-growth sites than between Late and Old-growth sites.

Within Age classes

The Morisita-Horn index values for community similarity between sites within the grouped age classes in the Picton is shown in Table 7.2. There were three obvious patterns.

(1) All taxa, except Tenebrionidae, had higher community similarity values in the Early sites than in the Intermediate sites.

(2) For all taxa, except Carabidae, community similarity was greater between Late regrowth sites than between Intermediate regrowth sites and

(3) for all taxa, except Carabidae, community similarity was greater between Late regrowth sites than between Old-growth sites. Community similarity values were particularly low between Old-growth sites for lucanids, staphylinids and tenebrionids.

Weilangta

Between Age classes

The Morisita-Horn index values for community composition between age classes also showed variation for many taxon in Weilangta, most markedly for lucanids (range 0 to 0.98) and staphylinids (range 0 to 0.87). However, there were several obvious patterns (Table 7.3).

(1) For all taxa, except for curculionids, community similarity between Early and Late was greater than between Early and Intermediate (the reverse of the trend shown in the Picton), although similarities in species composition between Early and Old-growth were for all taxa (similarly) greater than between Early and Late.

(2) For all taxa, except Staphylinidae, community similarity was greater between Intermediate and Old-growth than between Intermediate and Late.

(3) For all taxa except Curculionidae and Lucanidae, community similarity was greater between Late and Old-growth than between Intermediate and Old-growth.

Within Age classes

The Morisita-Horn index values for community similarity between sites within the grouped age classes in Weilangta is shown in Table 7.4. Several patterns were observed.

(1) For all taxa except Lucanidae, community similarity between the Early sites was greater than between the Intermediate sites.

(2) Community similarity values were higher or similar between Intermediate sites than Late except for staphylinids and tenebrionids, with no species in common between the two Intermediate sites.

(3) Community similarity values between Late sites were generally low and

(4) community similarity values between Old-growth sites were extremely variable for all taxa, except carabids.

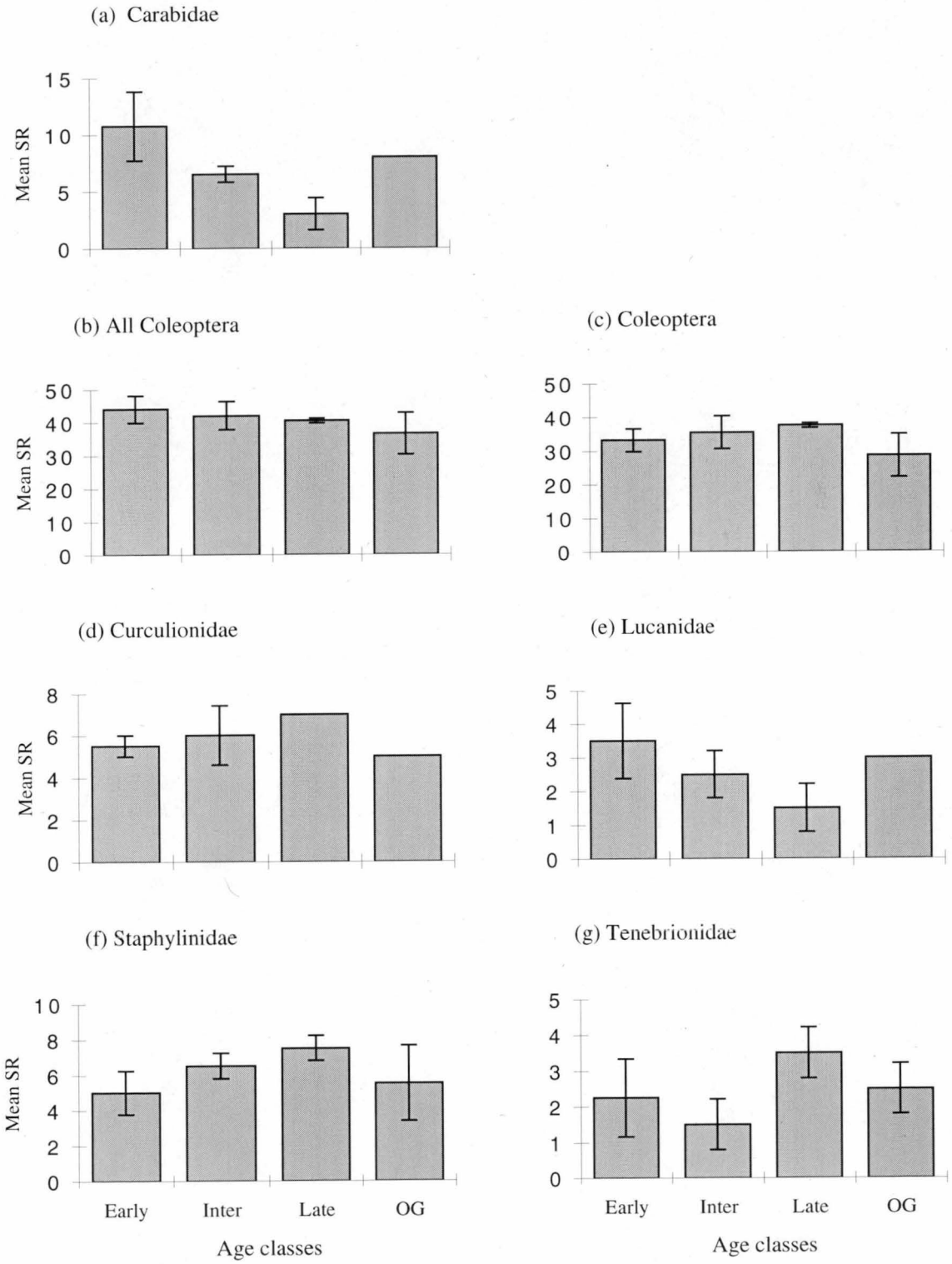


Figure 7.2. Mean species richness (SR) ± SD for each taxon within each forest age class in the Picton.

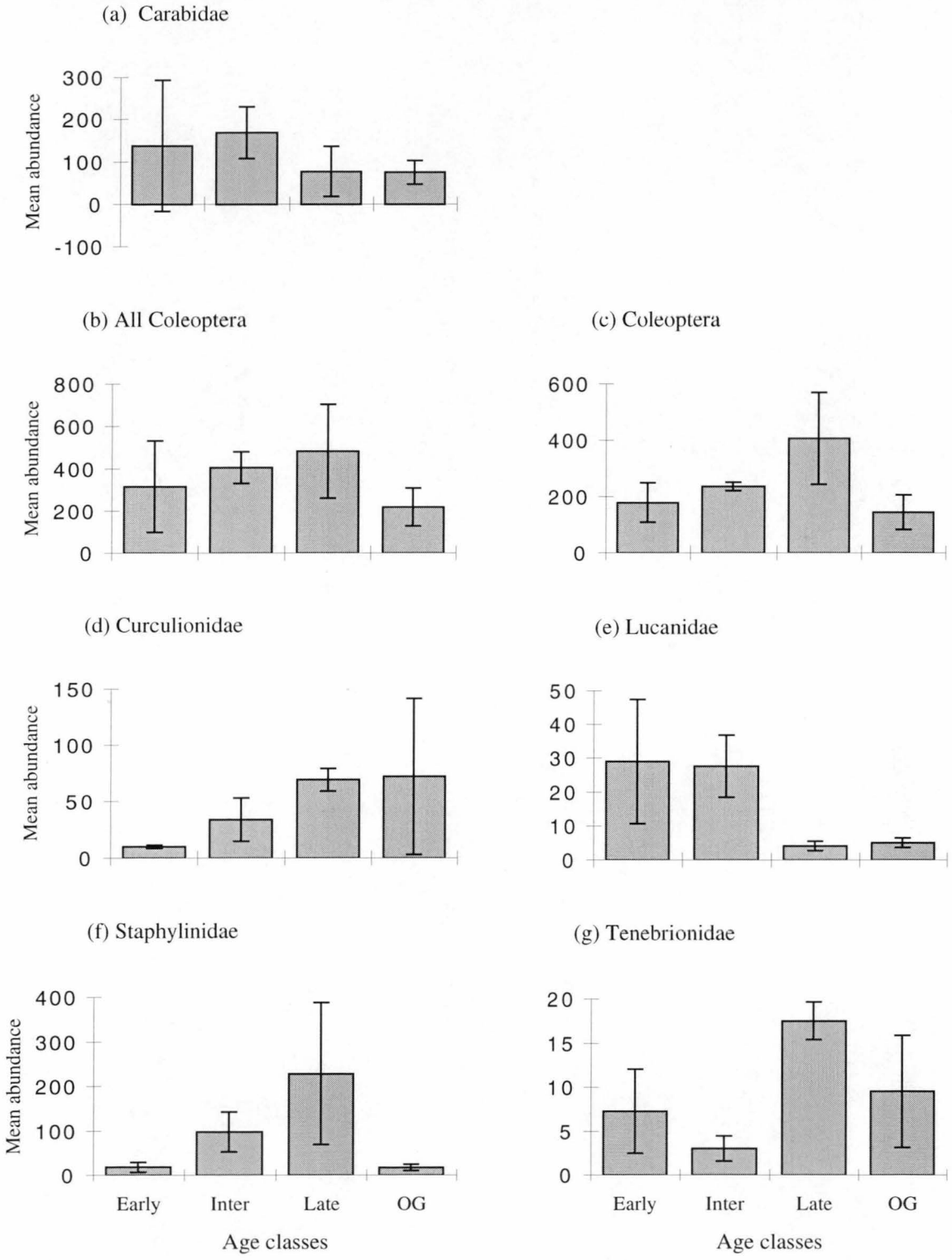


Figure 7.3. Mean abundance ± SD for each taxon within each forest age class in the Picton.

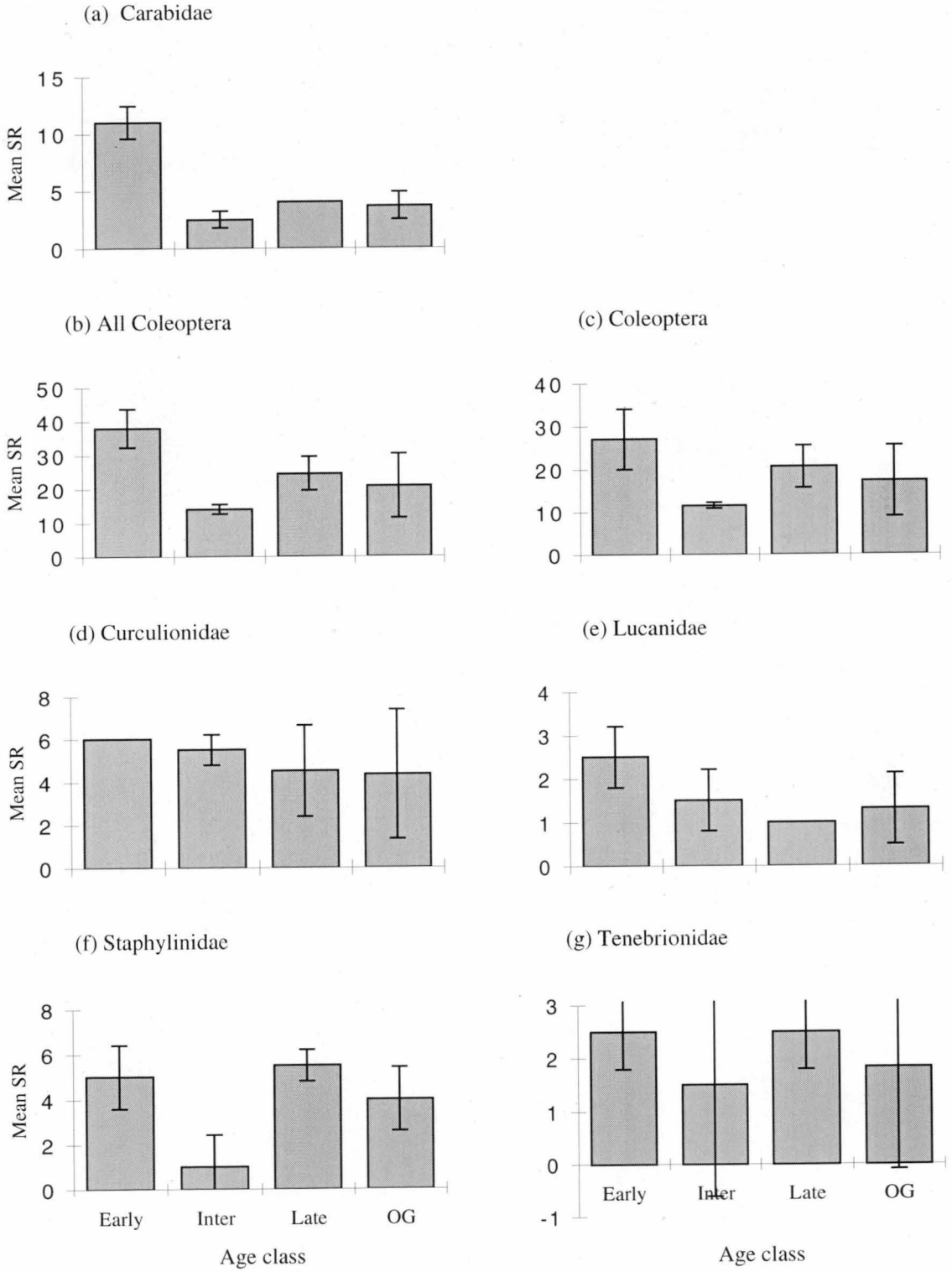


Figure 7.4. Mean species richness (SR) ± SD for each taxon within each forest age class in Weilangta.

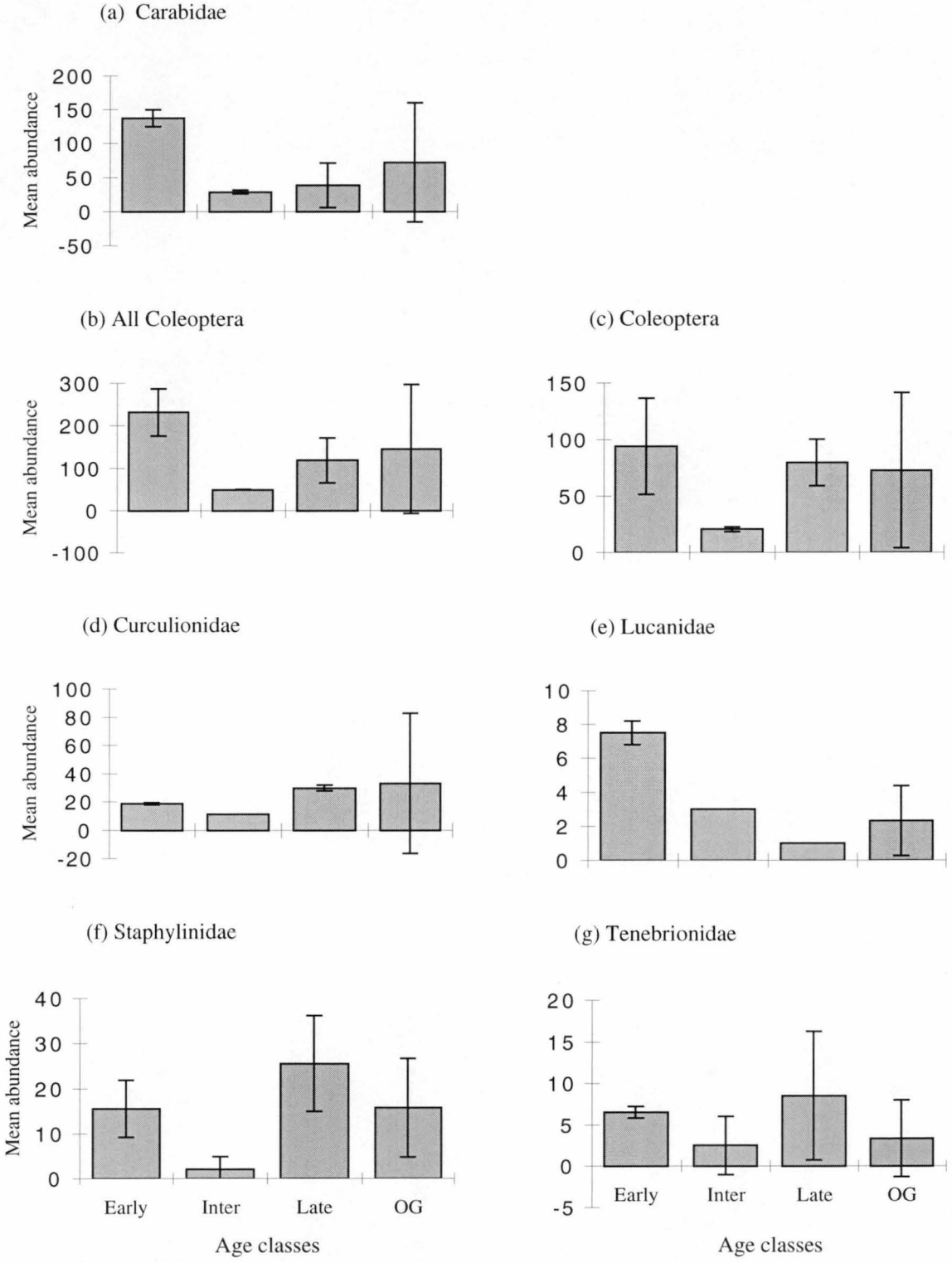


Figure 7.5. Mean abundance ± SD for each taxon within each forest age class in Weilangta.

TAXON Age Class	Early (1-4 years)	Intermediate (9years)	Late (25 years)
Carabidae			
Intermediate	0.53		
Late	0.33	0.87	
Old-growth	0.63	0.70	0.64
All Coleoptera			
Intermediate	0.64		
Late	0.57	0.67	
Old-growth	0.61	0.76	0.71
Coleoptera			
Intermediate	0.68		
Late	0 65	0.63	
Old-growth	0.62	0.78	0.73
Curculionidae			
Intermediate	0.87		
Late	0.87	0.91	
Old-growth	0.83	0.91	0.91
Lucanidae			
Intermediate	0.89		
Late	0.70	0.73	
Old-growth	0.86	0.94	0 67
Staphylinidae			
Intermediate	0.83		
Late	0.67	0.61	
Old-growth	0 81	0.68	0.82
Tenebrionidae			
Intermediate	0.85		
Late	0 63	0.86	
Old-growth	0.82	0.97	0.93

Table 7. 1. Morisita -Horn similarity values (max 1.0) for pairwise comparisons of similarity of community composition between forest age classes for each coleopteran taxon in the Picton.

TAXON	Age class	Early (1 year)	Early 4 years)	Age class	
Carabidae					
				Inter:Inter	0.95
	Early (1 yr)	0.76		Late:Late	0.72
	Early (4 yr)	0.87	0.50	OG: OG	0.82
All Coleoptera					
				Inter:Inter	0.84
	Early (1 yr)	0.73		Late:Late	0.88
	Early (4 yr)	0.66	0 62	OG: OG	0.65
Coleoptera					
				Inter:Inter	0 80
	Early (1 yr)	0.71		Late:Late	0.91
	Early (4 yr)	0.61	0.64	OG: OG	0.57
Curculionidae					
				Inter:Inter	0.79
	Early (1 yr)	0.50		Late:Late	0.98
	Early (4 yr)	0.88	0.75	OG: OG	0.86
Lucanidae					
				Inter:Inter	0.84
	Early (1 yr)	0.84		Late:Late	0.86
	Early (4 yr)	0.97	0.66	OG: OG	0.48
Staphylinidae					
				Inter:Inter	0.93
	Early (1 yr)	0.62		Late:Late	0.94
	Early (4 yr)	0.68	0.52	OG: OG	0.36
Tenebrionidae					
				Inter:Inter	0.00
	Early (1 yr)	0.47		Late:Late	0.89
	Early (4 yr)	0.69	0.67	OG: OG	0.24

Table 7. 2. Morisita -Horn similarity values (max 1.0) for pairwise comparisons of similarity of community composition within each forest age class for each coleopteran taxon in the Picton. Inter = Intermediate; OG = Old-growth.

TAXON Age Class	Early (1 year)	Intermediate (9years)	Late (20 years)
Carabidae			
Intermediate	0.49		
Late	0.56	0.56	
Old-growth	0.79	0.73	0.77
All Coleoptera			
Intermediate	0.46		
Late	0.48	0.43	
Old-growth	0.67	0.58	0.72
Coleoptera			
Intermediate	0.38		
Late	0.43	0.39	
Old-growth	0.58	0.50	0.70
Curculionidae			
Intermediate	0.69		
Late	0.55	0.50	
Old-growth	0.70	0.79	0 74
Lucanidae			
Intermediate	0 53		
Late	0.75	0.00	
Old-growth	0.98	0.67	0.63
Staphylinidae			
Intermediate	0.00		
Late	0.67	0.09	
Old-growth	0.87	0.00	0.87
Tenebrionidae			
Intermediate	0.30		
Late	0.47	0.60	
Old-growth	0.46	0.73	0.89

Table 7. 3. Morisita-Horn similarity values (max 1.0) for pairwise comparisons of similarity of community composition between forest age classes for each coleopteran taxon in Weilangta.

TAXON Age class			OG1	OG2	OG3	OG4	OG5
Carabidae							
		OG2	0.58				
Early:Early	0.85	OG3	0.50	0.87			
Inter:Inter	0.79	OG4	0.57	0.97	0.95		
Late:Late	0.45	OG5	0.41	0.80	0.88	0.87	
		OG6	0.47	0.85	0.76	0 85	0.85
All Coleoptera							
		OG2	0.49				
Early:Early	0.74	OG3	0.34	0.54			
Inter:Inter	0.66	OG4	0.36	0.59	0.75		
Late:Late	0.55	OG5	0.41	0.65	0 50	0.47	
		OG6	0.43	0.72	0.47	0.56	0.72
Coleoptera							
		OG2	0.43				
Early:Early	0.63	OG3	0.27	0.25			
Inter:Inter	0.50	OG4	0.24	0.25	0.42		
Late:Late	0.57	OG5	0.40	0.57	0.30	0.22	
		OG6	0 41	0.63	0.25	0.33	0.65
Curculionidae							
		OG2	0.33				
Early:Early	0.68	OG3	0.00	0 00			
Inter:Inter	0.64	OG4	0.48	0.17	0.00		
Late:Late	0.66	OG5	0.43	0.77	0.00	0.00	
		OG6	0.35	0.71	0.00	0.12	0.69
Lucanidae							
		OG2	0.00				
Early:Early	0.79	OG3	0.60	0.00			
Inter:Inter	1.00	OG4	0.60	0.00	1.00		
Late:Late	0.00	OG5	0.38	0.00	0.45	0.45	
		OG6	0.66	0.00	0.98	0.98	0.30
Staphylinidae							
		OG2	0.79				
Early:Early	0 52	OG3	0.66	0.39			
Inter:Inter	0 00	OG4	0.41	0.48	0.34		
Late:Late	0.67	OG5	0.76	0.68	0.84	0.38	
		OG6	0.82	0.84	0.62	0.60	0.75
Tenebrionidae							
		OG2	0.00				
Early:Early	0.83	OG3	0.00	0 00			
Inter:Inter	0.00	OG4	0.00	0.00	0.00		
Late:Late	0.40	OG5	0.57	0.00	0.00	0.73	
		OG6	0.44	0.00	0.00	0.57	0.84

Table 7. 4. Morisita -Horn similarity values (max 1.0) for pairwise comparisons of similarity of community composition within each forest age class for each Coleopteran taxon in Weilangta. Inter = Intermediate; OG = Old-growth.

SPECIES OCCURRENCE ACROSS THE SUCCESSIONAL GRADIENT

Classification of species

All species were classified according to their indicator values, determined by the INDVAL index, and their occurrence within each grouped class across the successional gradient. The number of individuals of each species, indicator values determined by the INDVAL index for each species and species classifications are shown in Tables 7.5. (Picton) and 7.6. (Weilangta). The characteristic (indicator) species (species with an indicator value of 25% or more) and symmetrical indicators (species with an indicator value >55%) for each age class are shown in Tables 7.7. (Picton) and 7.8. (Weilangta).

Species were classified as:

- (a) colonisers - defined as winged species and/or species generally restricted to and with their highest INDVAL in the Early successional stage;

- (b) successional specialists - defined as species generally occurring in regrowth sites, not or rarely occurring in the Old-growth sites, and with high INDVAL scores in either Intermediate or Late regrowth classes. These species were further classified as Intermediate or Late regrowth specialists on the basis of their INDVAL scores.

- (c) Forest generalists - defined as species occurring in all classes or all classes except for Early regrowth and

- (d) Old-growth specialists - defined as species predominantly restricted to Old-growth sites, or species with their only occurrence in Early sites, but unlikely to be colonisers (i.e. non-winged species).

The proportions of species occurring in each of these classifications and their occurrence in each of the forest classes are shown in Table 7.9. (Picton) and Table 7.10. (Weilangta). Species represented by a single individual were not included in these classifications, but their proportion and occurrence in each of the forest classes are given in the tables.

Picton

Only 9% of All Coleoptera were colonisers. Colonisers tended to be predominantly carabids. None of the other coleopteran taxon considered separately had species that could be classified as colonists.

One quarter (25%) of All Coleopteran species were successional specialists, and they comprised around 19% of the total coleopteran assemblage in the old-growth. Carabids had the lowest proportion of successional specialists (5%), Staphylinidae, the highest (50%). Curculionids had a greater proportion of Intermediate than Late successional specialists, while carabids and lucanids had no Late successional specialists at all. Staphylinids had a greater proportion of Late successional specialists, and tenebrionids had no Intermediate successional specialists.

Over a quarter (28%) of All Coleoptera species were forest generalists, and they comprised around a third of the species in the Early regrowth sites and around half of all species in each of the other forest classes. Half of all Curculionidae and Tenebrionidae species and one third of all lucanids and staphylinids were forest generalists. Carabids had the lowest proportion (14%). While the proportion of forest generalists in Early regrowth varied, they also comprised half or more of the species assemblages for all coleopteran taxon in each of the other forest classes. Carabids and lucanids had the highest proportion of forest generalists in Late regrowth, tenebrionids in Intermediate regrowth, curculionids in Old-growth, and staphylinid forest generalists comprised around half of the total species count in each of the forest classes.

A quarter (24%) of All Coleoptera species were Old-growth specialists and they made up 21% of the assemblage in Old-growth forest sites. Old-growth specialists comprised from 10 to 25% of all species for curculionids, lucanids, staphylinids and tenebrionids and formed around 11 to 17% of their respective assemblages in Old-growth forest sites. Interestingly, carabids had the highest proportion of Old-growth specialists (45%) and they formed 60% of the carabid assemblage in Old-growth forest sites.

14% of All Coleoptera species were represented by one individual. This proportion was similar for most other taxa, with two exceptions. Staphylinids had no species represented by single individuals, while around a third (33%) of all lucanid species recorded were single individuals, (illustrating the difficulty in trapping this cryptic fauna).

Weilangta

A similar proportion (9%) of All Coleoptera species in Weilangta were also classified as colonisers. Again, colonising species tended to be carabids and, with the exception of tenebrionids, none of the other coleopteran taxon considered separately had species that could be classified as colonists.

One third of All Coleopteran species were successional specialists and they made up around half of the species assemblages in Intermediate and Late regrowth sites and a third in Old-growth. Carabids had the lowest proportion of successional specialists (14%) (but no Intermediate successional specialists), Lucanidae and Staphylinidae the highest (both 67%). Staphylinids had a higher proportion of Late regrowth specialists while curculionids, lucanids and tenebrionids had equal numbers of Intermediate and Late regrowth specialists.

Only 13% of All Coleoptera species were classified as forest generalists, and they ranged from 19% (in Old-growth) to 48% of the species in each of the forest classes. 21% of carabid species were forest generalists. Lucanids and staphylinids had no forest generalist species. For all other taxa, the proportion of forest generalists varied from 23-29% and they comprised 25-40% of their respective Old-growth assemblages.

Over a third (33%) of All Coleoptera species could be classified as Old-growth specialists and they formed a similar proportion (39%) of the Old-growth assemblages. Both carabids and lucanids also had around a third of species designated as Old-growth specialists and comprising a similar proportion of their Old-growth assemblages. For all other taxa, less than a quarter (14 -23%) of their species could be classified as Old-growth specialists, although they formed from 20 -33% of Old-growth assemblages.

12% of All Coleoptera species were represented by one individual. This proportion was roughly similar for most other taxa, with two exceptions. Neither carabids nor lucanids had species represented by a single individual.

FAMILY	Species	Inds	Indicator Value (%) Early	Inter	Late	OG
EARLY COLONIZERS						
CARABIDAE	*Scopodes sigillatus*	**35**	67	24		5
CARABIDAE	*Mecyclothorax ambiguus*	**237**	74	42		
CARABIDAE	*Homothes elegans*	**15**	75			
CARABIDAE	*Scopodes atterimus*	**37**	100			
CARABIDAE	*Scopodes tasmanicus*	**4**	50			
CHRYSOMELIDAE	*Alticinae* sp. A	**6**	38			
CHRYSOMELIDAE	*Arsipoda* sp. A	**14**	32	35		
LYCIDAE	*Calachromus* sp. A	**4**	75			
MORDELLIDAE	*Mordella* sp A	**3**	75			
SUCCESSIONAL SPECIALISTS						
	Intermediate					
CARABIDAE	*Homothes guttifer*	**6**	13	50		
CHRYSOMELIDAE	*Chrysomelidae* sp. P/A	**3**	5	50		
CHRYSOMELIDAE	*Arsipoda* sp B	**7**	4	40	10	
CURCULIONIDAE	*Curculionidae* sp. P/E	**7**	8	50		
CURCULIONIDAE	*Curculionidae* sp. P/G	**2**		25	25	
LUCANIDAE	*Lissotes curvicornis*	**46**	23	90		10
LYCIDAE	*Metriorrhynchus* sp. A	**4**	17	25	25	
SILVANIDAE	*Cryptamorpha* sp. A	**3**		67		17
STAPHYLINIDAE	*Atheta* sp A	**27**		100		
	Late	**16**	12	4	16	8
CANTHARIDAE	*Heteromastix* sp. A	**5**	3		100	
CURCULIONIDAE	*Curculionidae* sp. P/I	**14**	20	6	88	
LATHRIDIIDAE	*Aridius nodifer*	**22**	68		50	
LEIODIDAE	*Eublackburniella* sp. A	**14**	1		85	8
LEIODIDAE	*Colon* sp. A	**15**	43	13	38	
PSELAPHIDAE	*Rybaxis* sp.A	**64**	32	2	96	
PSELAPHIDAE	*Rybaxis* sp.B	**82**	25		100	
PTILIDAE	*Rodwaya minutis*	**7**	42		100	
SCARABAEIDAE	*Heteronyx tasmanicus*	**2**			25	25
SCIRTIDAE	*Scirtidae* spp	**11**	8		38	13
SCYDMAENIDAE	*Euconnus* spp.	**5**	13		100	
STAPHYLINIDAE	*Heterothops* sp. A	**22**	1		71	14
STAPHYLINIDAE	*Anotylus* sp. A	**21**			86	7
STAPHYLINIDAE	*Anotylus* sp B	**331**	0	1	99	0
STAPHYLINIDAE	*Myllaena* sp. A	**4**			38	13
TENEBRIONIDAE	*Adelium foveatum*	**3**			33	17

Table 7.5. Indicator values, based on the INDVAL index, and species classifications, based on indicator values and occurrence in each of the classes across the successional gradient, for each species in the Picton. The no. of individuals for each species is indicated in bold.

FAMILY	Species	Inds	Indicator Value (%) Early	Inter	Late	OG
FOREST GENERALISTS						
(bold indicates class with highest indicator value for species)						
ANTHRIBIDAE	*Xnotropis micans*	**3**		17	17	17
CARABIDAE	*Sloaneana tasmaniae*	**167**	4	66	24	5
CARABIDAE	*Promecoderus tasmanicus*	**28**	10	53	13	21
CARABIDAE	*Rhabdotus reflexus*	**402**	5	59	28	13
CURCULIONIDAE	*Curculionidae* sp P/B	**52**	4	17	62	21
CURCULIONIDAE	*Curculionidae* sp. P/A	**93**	3	9	51	40
CURCULIONIDAE	*Curculionidae* sp P/H	**15**	8	17	17	25
CURCULIONIDAE	*Curculionidae* sp P/C	**94**	1	21	29	50
CURCULIONIDAE	*Curculionidae* sp. P/D	**105**	6	23	29	48
LEIODIDAE	*Nargomorphus* sp. B	**97**		72	1	27
LEIODIDAE	*Nargomorphus* sp A	**63**	16	24	58	18
LEIODIDAE	*Zeadolopus* spp	**75**	50	27	53	10
LEIODIDAE	*Nargomorphus* sp. C	**15**	1	18	11	43
LEIODIDAE	*Paragyrtodes* sp. A	**15**	4	23	14	14
LUCANIDAE	*Lissotes cancroides*	**143**	51	74	17	9
LUCANIDAE	*Lissotes rodwayi*	**27**	24	32	4	14
MELANDRYIDAE	*Orchesia* sp A	**21**	1	26	21	53
NITIDULIDAE	*Thalycrodes australe*	**19**	5	15	54	15
NITIDULIDAE	*Thalycrodes cylindricus*	**20**	4	3	88	3
PSELAPHIDAE	*Pselaphaulax* sp. A	**10**	1	33	56	6
SCARABAEIDAE	*Telura viticollis*	**11**	8	75	6	6
STAPHYLINIDAE	*Aleocharinae* sp. B	**142**	4	84	11	3
STAPHYLINIDAE	*Baeocera* spp	**31**	20	69	15	15
STAPHYLINIDAE	*Paederus* sp. A	**47**	3	53	44	1
STAPHYLINIDAE	*Aleocharinae* sp A	**120**	13	25	62	13
TENEBRIONIDAE	*Adelium abbreviatum*	**35**	9	4	58	17
TENEBRIONIDAE	*Adelium licinoides*	**26**		4	68	24
TENEBRIONIDAE	*Coripera deplanata*	**21**	50	14	7	29

Table 7.5. (Continued) Indicator values, based on the INDVAL index, and species classifications, based on indicator values and occurrence in each of the classes across the successional gradient, for each species in the Picton. The no. of individuals for each species is indicated in bold.

FAMILY	Species	Inds	Indicator Value (%) Early	Inter	Late	OG
OLD-GROWTH SPECIALISTS						
ANOBIIDAE	*Anobiidae* sp A	**5**	50	20		
ANTHICIDAE	*Formicus quadrimaculatus*	**5**	50			
BYRRHIDAE	*Microchaetes* spp.	**158**	71	13		38
BYRRHIDAE	*Pedilophorus* spp.	**6**	17	17		33
CARABIDAE	*Cyphotrechodes gibbipennis*	**5**	50			
CARABIDAE	*Sloanella* spp.	**4**	50			
CARABIDAE	*Pentagonica vittipennis*	**2**	25			
CARABIDAE	*Trechistus terricola*	**2**	25			
CARABIDAE	*Chylnus ater*	**40**	23			100
CARABIDAE	*Lestignathus cursor*	**3**	5	25		25
CARABIDAE	*Notonomus politulus*	**168**	59	2		95
CARABIDAE	*Percosoma carenoides*	**15**			27	47
CARABIDAE	*Prosopogmus tasmanicus*	**2**				50
CARABIDAE	*Stichonotus leai*	**9**	3			100
CURCULIONIDAE	*Curculionidae* sp P/F	**6**	17			50
ELATERIDAE	*Conoderus* sp B	**2**	50			
ELATERIDAE	*Conoderus* sp. A	**7**	4	40		60
LUCANIDAE	*Lissotes politus*	**11**	50			
OEDEMERIDAE	*Ischnomera* sp. A	**10**	50			
PSELAPHIDAE	*Mesoplatus* sp A	**3**	13	50		
STAPHYLINIDAE	*Scaphidium 4-pustulatum*	**2**	8	50		
STAPHYLINIDAE	*Tachinus* sp. A	**2**	25			
STAPHYLINIDAE	*Conosoma* sp A	**4**	30			50
TENEBRIONIDAE	*Promethis angulata*	**3**	50			
REPRESENTED BY ONE INDIVIDUAL						
CARABIDAE	*Amblystomus* spp.	**1**	1			
CARABIDAE	*Agonocheila curtula*	**1**	1			
CARABIDAE	*Trechimorphus diemenensis*	**1**		1		
CHRYSOMELIDAE	*Eurispa vitata*	**1**		1		
CHRYSOMELIDAE	*Monolepta* sp. A	**1**				1
CURCULIONIDAE	*Xyloborus* sp. A	**1**			1	
ELATERIDAE	*Agrypnus* sp. A	**1**				1
ELATERIDAE	*Conoderus* sp. C	**1**		1		
LATHRIDIIDAE	*Cortinicora* sp. A	**1**		1		
LUCANIDAE	*Ceratognathus niger*	**1**				1
LUCANIDAE	*Lissotes subcaeruleus*	**1**	1			
SCARABAEIDAE	*Saprosites* spp.	**1**			1	
TENEBRIONIDAE	*Brycopia* spp.	**1**			1	

Table 7.5. (Continued) Indicator values, based on the INDVAL index, and species classifications, based on indicator values and occurrence in each of the classes across the successional gradient, for each species in the Picton. The no. of individuals for each species is indicated in bold.

FAMILY	Species	Inds	Indicator Value (%) Early	Inter	Late	OG
EARLY COLONIZERS						
CARABIDAE	*Mecyclothorax ambiguus*	**22**	91		5	
CARABIDAE	*Scopodes aterrimus*	**20**	100			
CARABIDAE	*Scopodes sigillatus*	**12**	97			
CARABIDAE	*Hypharpax australis*	**2**	50			
CHRYSOMELIDAE	*Monolepta* spp.	**10**	50			
COCCINELLIDAE	*Coccinella undecimpunctata*	**6**	100			
TENEBRIONIDAE	*Atoichus bicolor*	**4**	100			
SUCCESSIONAL SPECIALISTS						
	Intermediate					
CURCULIONIDAE	*Curculionidae* sp. 9	**6**	13	67		11
CURCULIONIDAE	*Curculionidae* sp 1	**7**		69		5
ELATERIDAE	*Agrypnus* spp.	**2**	25	25		
LUCANIDAE	*Lissotes* sp. n	**19**	41	55		30
STAPHYLINIDAE	*Scaphidium 4-pustulatum*	**3**		33	17	
STAPHYLINIDAE	*Osoriinae* spp.	**2**		50		
TENEBRIONIDAE	*Adelium abbreviatum*	**3**		43		2
	Late					
CARABIDAE	*Homothes guttifer*	**5**	23		43	2
CARABIDAE	*Percosoma carenoides*	**37**	54		15	12
CHRYSOMELIDAE	*Altica* spp.	**2**	25		50	
CURCULIONIDAE	*Curculionidae* sp. 5	**28**	32		92	4
CURCULIONIDAE	*Curculionidae* sp 20	**28**			45	28
LATHRIDIIDAE	*Aridius nodifer*	**2**	25		50	
LEODIDAE	*Zeadolopus* spp.	**5**	17		25	8
LUCANIDAE	*Lissotes latidens*	**5**	27		30	13
OEDEMERIDAE	*Sessinia sublineatus*	**11**	91		50	
SCARABAEIDAE	*Heteronyx* sp. B	**12**			43	5
SCARABAEIDAE	*Heteronyx* sp. A	**2**			50	
STAPHYLINIDAE	*Oxytelus* spp.	**51**	6		79	14
STAPHYLINIDAE	*Quedius* spp.	**14**	9		81	10
STAPHYLINIDAE	*Conosoma* spp.	**11**	30		60	13
STAPHYLINIDAE	*Atheta* spp	**45**	30		53	47
TENEBRIONIDAE	*Adelium licinoides*	**11**	12		79	11

Table 7.6. Indicator values, based on the INDVAL index, and species classifications, based on indicator values and occurrence in each of the classes across the successional gradient, for each species in Weilangta. The no. of individuals for each species is indicated in bold.

FAMILY	Species	Inds	Indicator Value (%) Early	Inter	Late	OG
OLD-GROWTH SPECIALISTS						
BYRRHIDAE	*Mıchrochaetes* spp.	**35**	96	1		4
CARABIDAE	*Eutrechus* spp	**7**	50			
CARABIDAE	*Amblystomus nıger*	**2**	50			
CARABIDAE	*Rhabdotus floridus*	**50**	83			17
CARABIDAE	*Sloaneana tasmaniae*	**29**	78	11		13
CARABIDAE	*Sımodontus australıs*	**5**	46			17
COLYDIIDAE	*Colydiidae* spp	**5**			21	29
CURCULIONIDAE	*Curculionidae* sp. 2	**17**	73	20		4
CURCULIONIDAE	*Curculıonıdae* sp. 14	**17**	51	13		24
CURCULIONIDAE	*Curculionıdae* sp. 3	**8**	38			33
CURCULIONIDAE	*Curculionidae* sp. 10	**3**				33
CURCULIONIDAE	*Curculionidae* sp 8	**2**				17
ELATERIDAE	*Conoderus* spp.	**15**	33			50
ELATERIDAE	*Elatichrosus* spp.	**6**				33
HYDROPHILIDAE	*Cercyon* spp.	**38**				17
LEODIDAE	*Leodıdae* sp. C	**4**	100			
LEODIDAE	*Nargomorphus* spp.	**12**	23		14	36
LEODIDAE	*Eublackburniella* spp.	**4**	25			17
LUCANIDAE	*Lissotes obtusatus*	**13**	72		19	42
MELANDRYIDAE	*Orchesia* spp.	**7**				17
PHLOEOSTICHIDAE	*Hymaea succınıfera*	**4**	50			
PSELAPHIDAE	*Pselaphaulax* spp.	**3**	100			
PSELAPHIDAE	*Rybaxıs* spp.	**7**	47			17
SCARABAEIDAE	*Adoryphorus couloni*	**2**				17
SCARABAEIDAE	*Telura vittıcollis*	**2**				33
STAPHYLINIDAE	*Anotylus* spp.	**29**	4		33	56
STAPHYLINIDAE	*Staphylininae* spp	**24**	75			67
TENEBRIONIDAE	*Promethıs angulata*	**2**				17
TROGIDAE	*Omorgus australasıae*	**2**				17

Table 7.6. (Continued) Indicator values, based on the INDVAL index, and species classifications, based on indicator values and occurrence in each of the classes across the successional gradient, for each species in Weilangta. The no. of individuals for each species is indicated in bold.

FAMILY	Species	Inds	Indicator Value (%) Early	Inter	Late	OG
FOREST GENERALISTS						
(bold indicates class with highest indicator value for species)						
CARABIDAE	*Promecoderus brunnicornis*	**313**	29	2	46	**51**
CARABIDAE	*Notonomus politulus*	**303**	46	47	2	**51**
CARABIDAE	*Chylnus ater*	**29**	11	7	**27**	22
CURCULIONIDAE	*Curculionidae* sp 4	**149**	2	10	**41**	33
CURCULIONIDAE	*Curculionidae* sp. 19	**37**	16	6	16	9
CURCULIONIDAE	*Curculionidae* sp. 11	**12**	8	18	9	**24**
NITIDULIDAE	*Thalycrodes australe*	**17**	8	29	**48**	11
SCARABAEIDAE	*Onthophagus fuliginosus*	**14**		5	**70**	10
TENEBRIONIDAE	*Isopteron AD 47*	**21**		7	**31**	12
TENEBRIONIDAE	*Coripera deplanata*	**13**	64	13	**25**	13
REPRESENTED BY ONE INDIVIDUAL						
CHRYSOMELIDAE	*Paropsis* spp.	**1**				17
CURCULIONIDAE	*Curculionidae* sp 7	**1**			50	
LEODIDAE	*Pseudonemadus* spp.	**1**				17
MORDELLIDAE	*Mordella* spp	**1**			50	
SCARABAEIDAE	*Heteronyx* sp D	**1**	50			
SCARABAEIDAE	*Scitala sericans*	**1**				17
SCARABAEIDAE	*Onthophagus pronus*	**1**			50	
STAPHYLINIDAE	*Paederinae* spp.	**1**	50			
TENEBRIONIDAE	*Lepispilus sulcicollis*	**1**			50	

Table 7.6. (Continued) Indicator values, based on the INDVAL index, and species classifications, based on indicator values and occurrence in each of the classes across the successional gradient, for each species in Weilangta. The no of individuals for each species is indicated in bold.

Early	Intermediate	Late	Old-growth
(Crb) *Scopodes sigillatus* (67)	(Crb) *Homothes guttifer* (50)	(Can) *Heteromastix* sp A (100)	(Byr) *Microchaetes* spp. (38)
(Crb) *Mecyclothorax ambiguus* (74)	(Crb) *Sloaneana tasmaniae* (66)	(Crb) *Rhabdotus reflexus* (28)	(Byr) *Pedalophorus* spp. (33)
(Crb) *Homothes elegans* (75)	(Crb) *Mecyclothorax ambiguus* (50)	(Crb) *Percosoma carenoides* (27)	(Crb) *Chylnus ater* (100)
(Crb) *Scopodes atterimus* (100)	(Crb) *Promecoderus tasmanicus* (53)	(Curc) *Curculionidae* sp P/B (62)	(Crb) *Lestignathus cursor* (25)
(Crb) *Scopodes tasmanicus* (50)	**(Crb) *Rhabdotus reflexus* (59)**	**(Curc) *Curculionidae* sp. P/A (51)**	(Crb) *Notonomus politus* (95)
(Chr) *Alticinae* sp A (38)	**(Crb) *Scopodes sigillatus* (71)**	(Curc) *Curculionidae* sp P/C (29)	**(Crb) *Percosoma carenoides* (47)**
(Chr) *Arsipoda* sp. A (32)	(Chr) *Chrysomelidae* sp P/A (50)	(Curc) *Curculionidae* sp P/D (29)	(Crb) *Prosopogmus tasmanicus* (50)
(Lyc) *Calachromus* sp A (75)	(Chr) *Arsipoda* sp B (40)	(Lat) *Aridius nodifer* (50)	(Crb) *Stichonotus leai* (100)
(Mor) *Mordella* sp A (75)	(Curc) *Curculionidae* sp P/E (50)	(Leio) *Eublackburniella* sp A (85)	(Curc) *Curculionidae* sp P/F (50)
	(Elat) *Conoderus* sp A (40)	(Leio) *Nargomorphus* sp A (58)	**(Curc) *Curculionidae* sp. P/C (50)**
	(Leio) *Nargomorphus* sp. B (72)	**(Leio) *Zeadolopus* spp. (53)**	**(Curc) *Curculionidae* sp. P/A (40)**
	(Leio) *Zeadolopus* spp (27)	(Nit) *Thalycrodes australe* (54)	**(Curc) *Curculionidae* sp. P/H (25)**
	(Luc) *Lissotes curvicornis* (90)	(Nit) *Thalycrodes cylindricus* (88)	**(Curc) *Curculionidae* sp. P/D (48)**
	(Luc) *Lissotes cancroides* (74)	(Psel) *Rybaxis* sp A (96)	**(Elat) *Conoderus* sp. A (60)**
	(Luc) *Lissotes rodwayi* (32)	(Psel) *Rybaxis* sp B (100)	(Leio) *Nargomorphus* sp B (27)
	(Mel) *Orchesia* sp A (26)	**(Psel) *Pselaphaulax* sp. A (56)**	**(Leio) *Nargomorphus* sp. C (43)**
	(Psel) *Pselaphaulax* sp A (33)	(Ptil) *Rodwaya minutis* (100)	(Luc) *Lissotes politus* (50)
	(Scrb) *Telura viticollis* (74)	(Scyd) *Euconnus* spp (100)	**(Mel) *Orchesia* sp. A (53)**
	(Sil) *Cryptamorpha* sp A (67)	(Stap) *Heterothops* sp A (71)	(Stap) *Conosoma* sp A (50)
	(Stap) *Atheta* sp A (100)	(Stap) *Anotylus* sp A (86)	(Ten) *Coripera deplanata* (29)
	(Stap) *Aleocharinae* sp B (84)	(Stap) *Anotylus* sp B (99)	
	(Stap) *Baeocera* spp (69)	**(Stap) *Aleocharinae* sp. A (62)**	
	(Stap) *Paederus* sp A (53)	(Ten) *Adelium abbreviatum* (58)	
	(Stap) *Aleocharinae* sp A (25)	(Ten) *Adelium licinoides* (68)	
		Curc *Curculionidae* sp P/I (88)	

Table 7.7. Characteristic (indicator) species for each forest class in the Picton, showing the species indicator value in parentheses. All species with an indicator value > 25% are shown for each age class where they are found. The class where a species has the maximum indicator value is indicated in bold. Species with an indicator value >55% are symmetrical indicators (*sensu* Dufrene and Legendre 1997).

Early		Intermediate		Late		Old-growth	
(Crb)	***Mecyclothorax ambiguus* (91)**	(Byr)	*Mıchrochaetes* spp (38)	(Crb)	*Homothes guttıfer (43)*	(Col)	*Colydiıdae* spp (29)
(Crb)	*Scopodes aterrimus* (100)	(Curc)	*Curculionidae* sp 9 (67)	(Crb)	Mecyclothorax ambıguus *(50)*	(Crb)	*Promecoderus brunnıcornıs* (51)
(Crb)	*Scopodes sıgıllatus* (97)	(Curc)	*Curculıonıdae* sp 2 (75)	(Curc)	*Curculıonıdae* sp 5 (92)	(Crb)	*Notonomus polıtulus* (51)
(Crb)	*Hypharpax australis* (50)	(Curc)	*Curculıonıdae* sp 14 (26)	**(Curc)**	***Curculionidae* sp. 20 (45)**	(Crb)	*Rhabdotus floridus*
(Chr)	*Monolepta* spp (50)	(Curc)	*Curculıonidae* sp 1 (69)	(Lat)	*Aridius nodıfer* (50)	(Crb)	*Sloaneana tasmanıae*
(Coc)	*Coccınella undecımpunctata* (100)	(Elat)	*Agrypnus* spp (50)	(Leıo)	*Zeadolopus* spp (25)	(Crb)	*Simodontus australıs*
(Ten)	*Atoıchus bıcolor* (100)	**(Luc)**	***Lissotes* sp. n (55)**	(Luc)	*Lıssotes latıdens* (30)	(Curc)	*Curculıonıd* sp 4 (33)
		(Stap)	*Osorıınae* spp (50)	(Oed)	*Sessınıa sublınıatus* (50)	(Curc)	*Curculıonıd* sp. 20 (28)
		(Ten)	*Adelium abbreviatum* (43)	(Scrb)	*Heteronyx* sp B (43)	(Curc)	*Curculıonıd* sp 3 (33)
		(Crb)	*Notonomus polıtulus* (47)	(Scrb)	*Heteronyx* sp A (50)	(Curc)	*Curculionid* sp 10 (33)
		(Nıt)	*Thalycrodes australe* (29)	(Stap)	*Oxytelus* spp (79)	(Elat)	*Conoderus* spp (50)
				(Stap)	*Quedius* spp (81)	(Elat)	*Elatıchrosus* spp (50)
				(Stap)	*Conosoma* spp (60)	(Leıo)	*Nargomorphus* spp (36)
				(Stap)	***Atheta* spp. (53)**	(Luc)	*Lissotes* sp n (30)
				(Ten)	*Adelium lıcınoıdes* (79)	(Luc)	*Lıssotes obtusatus* (42)
				(Crb)	*Promecoderus brunnıcornıs* (46)	(Scrb)	*Telura vıttıcollis* (33)
				(Crb)	*Chylnus ater* (27)	(Stap)	*Atheta* spp (47)
				(Curc)	*Curculıonıdae* sp 4 (41)	**(Stap)**	***Anotylus* spp. (56)**
				(Nıt)	*Thalycrodes australe* (48)	(Stap)	*Staphylininae* spp (67)
				(Scrb)	*Onthophagus fulıgınosus* (70)		
				(Ten)	*Isopteron AD 47* (31)		
				(Ten)	*Corıpera deplanata* (25)		
				(Stap)	*Anotylus* spp (33)		

Table 7.8. Characteristic (indicator) species for each forest class in Weilangta, showing the species indicator value in parentheses. All species with an indicator value > 25% are shown for each age class where they are found The class where a species has the maximum indicator value is indicated in bold. Species with an indicator value >55% are symmetrical indicators (*sensu* Dufrene and Legendre 1997).

Carabidae

Forest Class	All	Early	Intermediate	Late	Old-growth
Total number of species	22	19	9	4	10
Total catch	1184	548	336	152	148
Colonizers	23%	26%	22%	0%	10%
Sucessional specialists	5%	5%	11%	0%	0%
Successional specialists: INTER	5%	5%	11%	0%	0%
Successional specialists:LATE	0%	0%	0%	0%	0%
Forest generalists	14%	16%	33%	75%	30%
Old-growth specialists	45%	42%	22%	25%	60%
1 individual	14%	11%	11%	0%	0%

All Coleoptera

Forest Class	All	Early	Intermediate	Late	Old-growth
Total number of species	99	77	57	51	53
Total catch	3466	1260	807	964	435
Colonizers	9%	12%	7%	0%	2%
Sucessional specialists	25%	24%	23%	37%	19%
Successional specialists: INTER	9%	8%	16%	6%	4%
Successional specialists.LATE	16%	16%	7%	31%	15%
Forest generalists	28%	32%	49%	55%	53%
Old-growth specialists	24%	29%	14%	2%	21%
1 individual	13%	4%	7%	6%	6%

Curculionidae

Forest Class	All	Early	Intermediate	Late	Old-growth
Total number of species	10	8	8	8	6
Total catch	389	40	67	138	144
Colonizers	0%	0%	0%	0%	0%
Sucessional specialists	30%	25%	38%	25%	0%
Successional specialists: INTER	20%	13%	25%	13%	0%
Successional specialists·LATE	10%	13%	13%	13%	0%
Forest generalists	50%	63%	63%	63%	83%
Old-growth specialists	10%	13%	0%	0%	17%
1 individual	10%	0%	0%	13%	0%

Table 7.9. Total number of species and individuals and proportion (%) of species in classification categories, overall, and in each forest class, for each taxon, in the Picton.

Lucanidae

Forest Class	All	Early	Intermediate	Late	Old-growth
Total number of species	6	5	3	2	4
Total catch	229	116	55	8	10
Colonizers	0%	0%	0%	0%	0%
Sucessional specialists	17%	20%	33%	0%	25%
Successional specialists: INTER	17%	20%	33%	0%	25%
Successional specialists.LATE	0%	0%	0%	0%	0%
Forest generalists	33%	40%	67%	100%	50%
Old-growth specialists	17%	20%	0%	0%	0%
1 individual	33%	20%	0%	0%	25%

Staphylinidae

Forest Class	All	Early	Intermediate	Late	Old-growth
Total number of species	12	9	7	8	9
Total catch	753	70	194	455	34
Colonizers	0%	0%	0%	0%	0%
Sucessional specialists	50%	33%	43%	50%	44%
Successional specialists. INTER	8%	0%	14%	0%	0%
Successional specialists:LATE	33%	22%	14%	50%	44%
Forest generalists	33%	44%	57%	50%	44%
Old-growth specialists	25%	33%	14%	0%	11%
1 individual	0%	0%	0%	0%	0%

Tenebrionidae

Forest Class	All	Early	Intermediate	Late	Old-growth
Total number of species	6	3	3	5	4
Total catch	89	29	6	35	19
Colonizers	0%	0%	0%	0%	0%
Sucessional specialists	17%	0%	0%	20%	25%
Successional specialists: INTER	0%	0%	0%	0%	0%
Successional specialists:LATE	17%	0%	0%	20%	25%
Forest generalists	50%	67%	100%	60%	75%
Old-growth specialists	17%	33%	0%	0%	0%
1 individual	17%	0%	0%	20%	0%

Table 7.9. (Continued) Total number of species and individuals and proportion (%) of species in classification categories, overall, and in each forest class, for each taxon, in the Picton.

Carabidae

	All	Early	Intermediate	Late	Old-growth
Total number of species	14	14	4	6	9
Total catch	836	274	56	76	430
Colonizers	29%	29%	0%	17%	11%
Sucessional specialists	14%	14%	0%	33%	22%
Successional specialists: INTER	0%	0%	0%	0%	0%
Successional specialists:LATE	14%	14%	0%	33%	22%
Forest generalists	21%	21%	75%	50%	33%
Old-growth specialists	36%	36%	25%	0%	33%
1 individual	0%	0%	0%	0%	0%

All Coleoptera

	All	Early	Intermediate	Late	Old-growth
Total number of species	78	52	21	36	54
Total catch	1658	462	97	235	864
Colonizers	9%	13%	0%	3%	2%
Sucessional specialists	33%	37%	48%	47%	35%
Successional specialists: INTER	9%	6%	33%	3%	7%
Successional specialists:LATE	21%	25%	0%	44%	22%
Forest generalists	13%	15%	48%	28%	19%
Old-growth specialists	37%	37%	19%	11%	44%
1 individual	12%	4%	0%	11%	6%

Curculionidae

	All	Early	Intermediate	Late	Old-growth
Total number of species	13	8	7	6	12
Total catch	315	37	22	59	197
Colonizers					
Sucessional specialists	46%	50%	57%	33%	50%
Successional specialists: INTER	15%	13%	29%	0%	17%
Successional specialists:LATE	15%	13%	0%	33%	17%
Forest generalists	23%	38%	43%	50%	25%
Old-growth specialists	38%	38%	29%	0%	42%
1 individual	8%	0%	0%	17%	0%

Table 7.10. Total number of species and individuals and proportion (%) of species in classification categories, overall, and in each forest class, for each taxon, in Weilangta.

Lucanidae

	All	Early	Intermediate	Late	Old-growth
Total number of species	3	3	1	2	3
Total catch	37	15	6	2	14
Colonizers					
Sucessional specialists	67%	67%	100%	50%	67%
Successional specialists: INTER	33%	33%	100%		33%
Successional specialists:LATE	33%	33%	0%	50%	33%
Forest generalists	0%	0%	0%	0%	0%
Old-growth specialists	33%	33%	0%	50%	33%
1 individual	0%	0%	0%	0%	0%

Staphylinidae

	All	Early	Intermediate	Late	Old-growth
Total number of species	9	7	2	6	6
Total catch	180	31	4	51	94
Colonizers					
Sucessional specialists	67%	57%	100%	83%	67%
Successional specialists: INTER	22%	0%	100%	17%	0%
Successional specialists:LATE	44%	57%	0%	67%	67%
Forest generalists	0%	0%	0%	0%	0%
Old-growth specialists	22%	29%	0%	17%	33%
1 individual	11%	14%	0%	0%	0%

Tenebrionidae

	All	Early	Intermediate	Late	Old-growth
Total number of species	7	3	3	4	5
Total catch	55	13	5	17	20
Colonizers	14%	33%	0%	0%	0%
Sucessional specialists	29%	33%	33%	25%	40%
Successional specialists: INTER	14%	0%	33%	0%	20%
Successional specialists:LATE	14%	33%	0%	25%	20%
Forest generalists	29%	33%	67%	50%	40%
Old-growth specialists	14%	0%	0%	0%	20%
1 individual	14%	0%	0%	25%	0%

Table 7.10. (Continued) Total number of species and individuals and proportion (%) of species in classification categories, overall, and in each forest class, for each taxon, in Weilangta.

UPGMA cluster analysis

Picton

The results of the UPGMA cluster analysis for carabids, in the Picton (Fig. 7.6.), showed that although the carabid assemblages of the regrowth sites were different from those in Old-growth, carabid assemblages in Intermediate and Late regrowth classes were similar. The UPGMA analysis for Coleoptera (Fig. 7.8.) gave the same result as the analysis based on All Coleoptera (Fig. 7.7.). Although results for both showed clear separation of regrowth classes with time since clearfelling, this did not apply to Old-growth sites. While clustering separated sites according to their regrowth age, Old-growth sites were grouped with Intermediate regrowth sites, indicating similarities in species composition. The results of the UPGMA analysis for staphylinids (Fig. 7.11.) were somewhat similar, except that only one Old-growth site clustered with the Intermediate sites, while the other clustered with Early regrowth sites.

There was some similarity in the patterns of site groupings of carabids and All Coleoptera (and Coleoptera), in that age classes consistently clustered together for both. However, in the cluster analysis based on carabid data, Old-growth sites grouped with an Early site, and the Intermediate grouped with the Late regeneration class sites; while in the cluster analysis based on All Coleoptera data, Old-growth sites grouped with Intermediate regeneration class sites and the Late regeneration class sites formed a separate cluster. The Mantel tests showed there was a significant positive correlation between the UPGMA matrices for pairwise comparisons of carabids and All Coleoptera, Coleoptera and Curculionidae, but no significant similarity in the patterns of site groupings between carabids and all other coleopteran taxa, or between pairwise comparisons of All Coleoptera, Coleoptera and other coleopteran taxa, although some sites frequently grouped together.

Results of the UPGMA analyses for lucanids (Fig. 7.10.) showed a clear distinction between Intermediate and Late regrowth species assemblages, but showed both assemblages contained species occurring in Early regrowth sites. Patterns of similarity of assemblages produced by the cluster analysis for tenebrionids (Fig. 7.12.) showed little evidence of relationship to forest class, except for the grouping of the two Late regrowth sites; while results suggest that curculionid species (Fig. 7.9.) discriminated only between Early regrowth and all other age classes.

Picton Carabidae

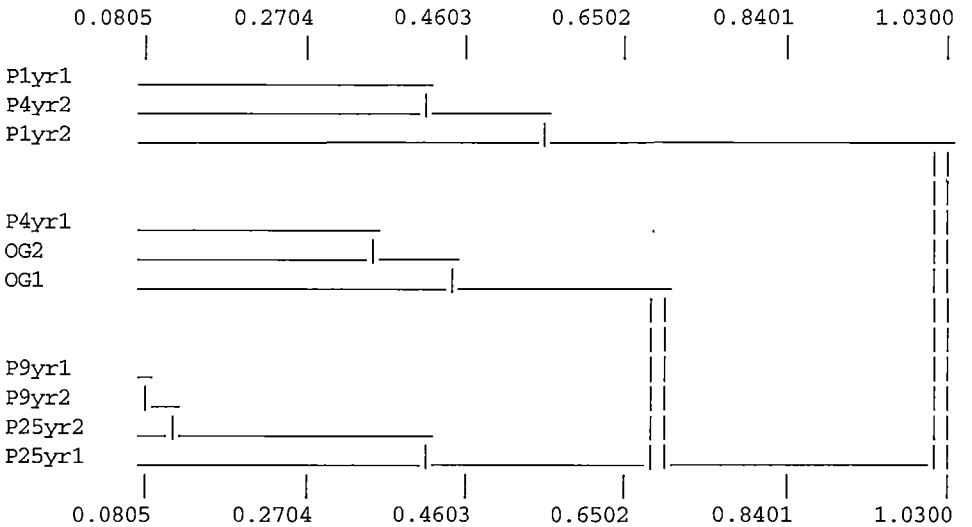


Figure 7.6. Dendrogram from UPGMA analysis, based on the species composition of the carabid beetle fauna in the Picton forest, showing the degree of similarity between sites (|) within the three clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Details relating to the site codes used here are given in Chapter Four.

Picton All Coleoptera

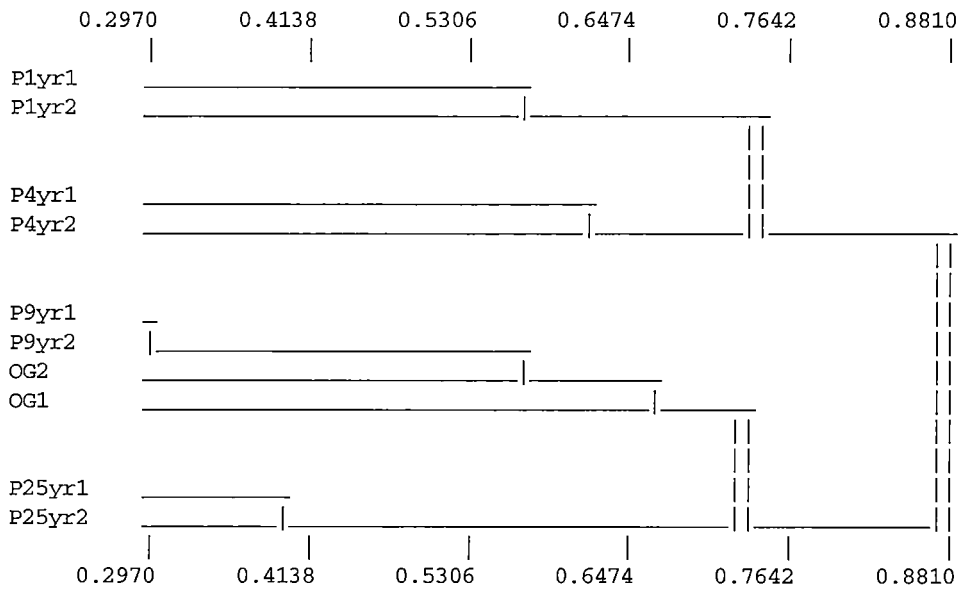


Figure 7.7. Dendrogram from UPGMA analysis, based on the species composition of the total ground-dwelling coleopteran fauna in the Picton forest, showing the degree of similarity between sites (|) within the four clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Details relating to the site codes used here are given in Chapter Four.

Picton Coleoptera

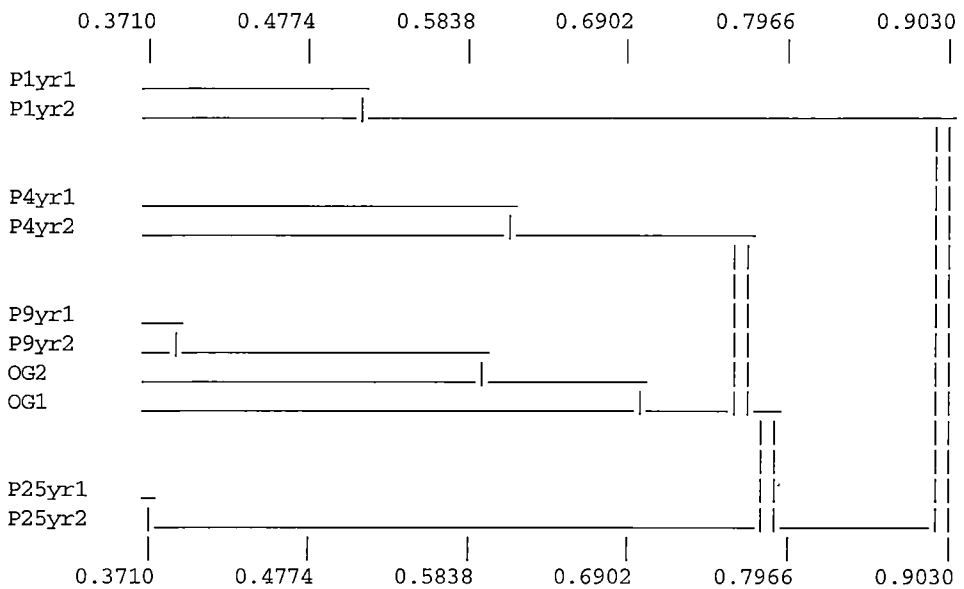


Figure 7.8. Dendrogram from UPGMA analysis, based on the species composition of the ground-dwelling coleopteran fauna (excluding carabids) in the Picton forest, showing the degree of similarity between sites (|) within the four clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Details relating to the site codes used here are given in Chapter Four.

Picton Curculionidae

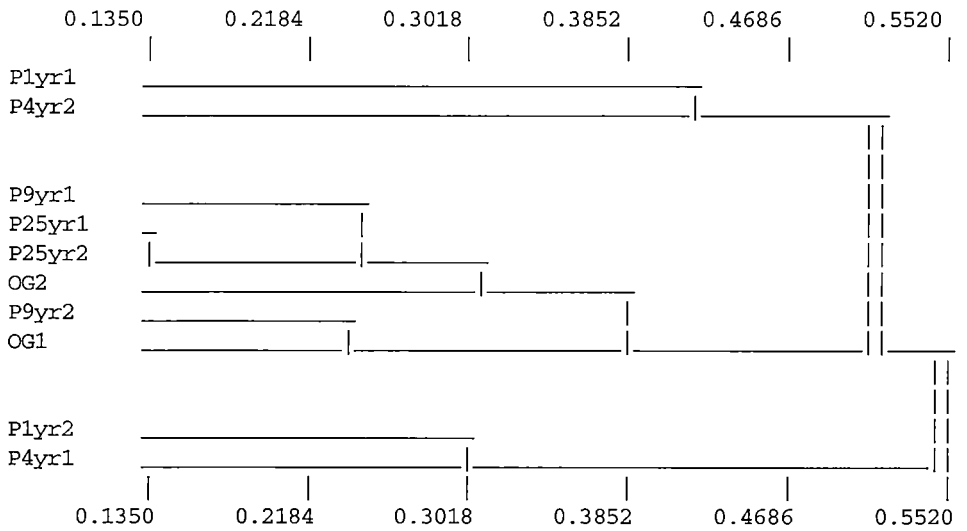


Figure 7.9. Dendrogram from UPGMA analysis, based on the species composition of the curculionid fauna in the Picton forest, showing the degree of similarity between sites (|) within the three clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Details relating to the site codes used here are given in Chapter Four.

Picton Lucanidae

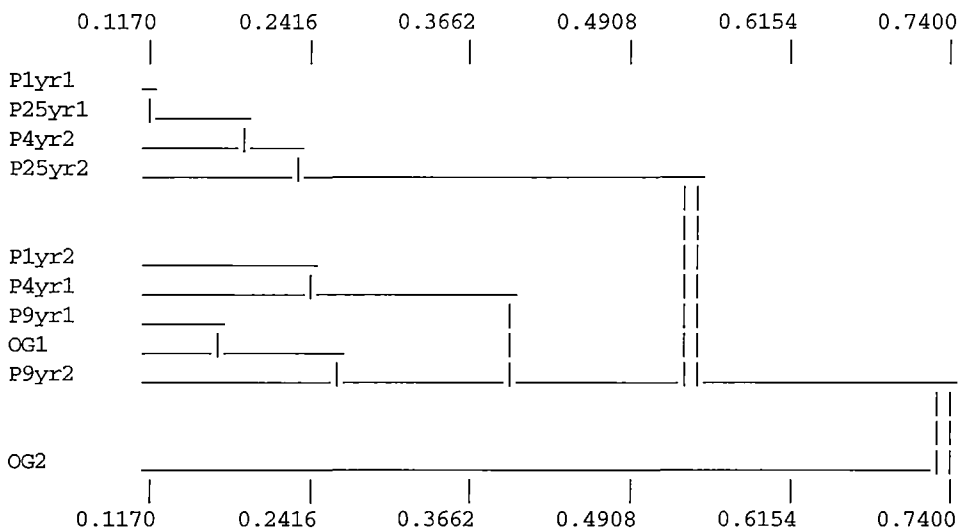


Figure 7.10. Dendrogram from UPGMA analysis, based on the species composition of the lucanid fauna in the Picton forest, showing the degree of similarity between sites (|) within the three clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Details relating to the site codes used here are given in Chapter Four.

Picton Staphylinidae

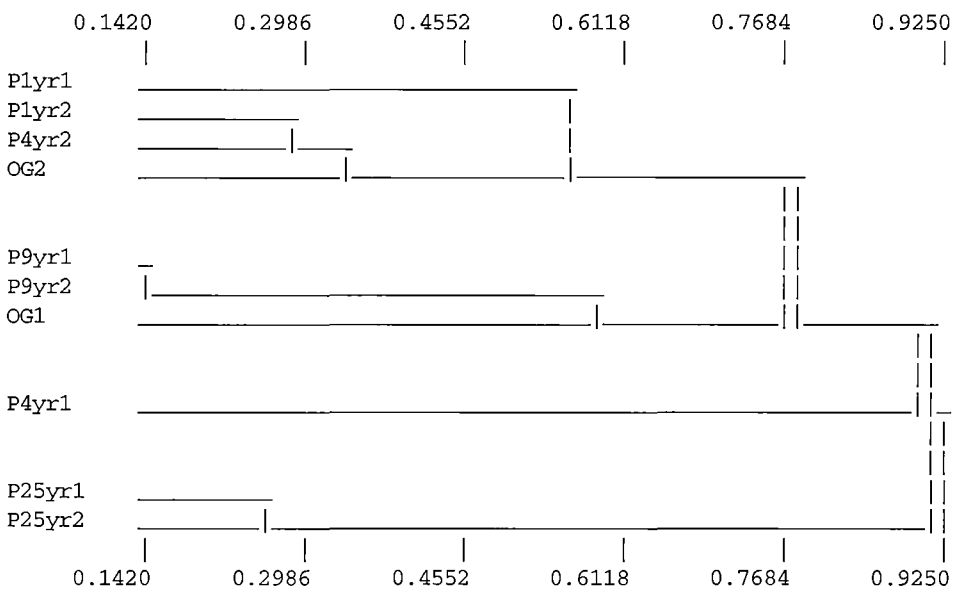


Figure 7.11 Dendrogram from UPGMA analysis, based on the species composition of the staphylinid fauna in the Picton forest, showing the degree of similarity between sites (|) within the four clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Details relating to the site codes used here are given in Chapter Four.

Picton Tenebrionidae

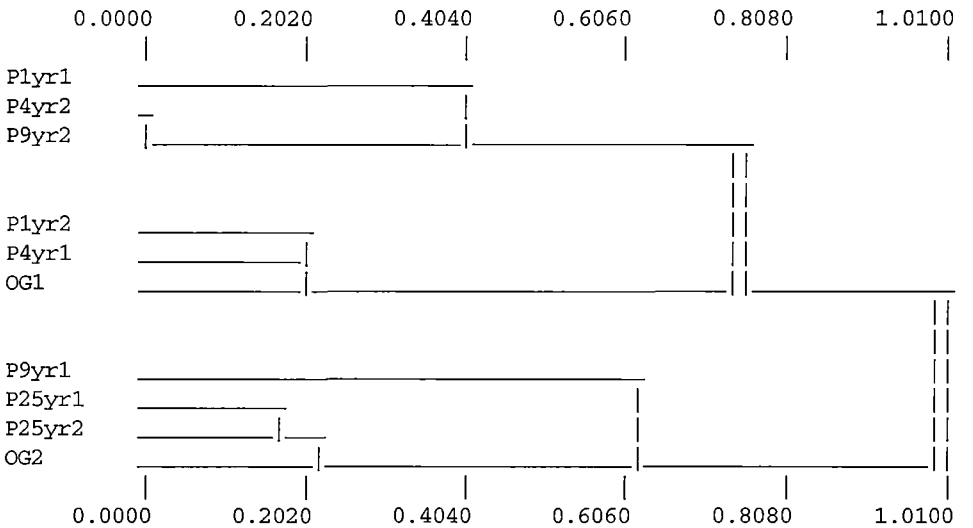


Figure 7.12. Dendrogram from UPGMA analysis, based on the species composition of the tenebrionid fauna in the Picton forest, showing the degree of similarity between sites (|) within the three clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Details relating to the site codes used here are given in Chapter Four.

Weilangta

Cluster analyses for beetles in Weilangta indicated that the composition of carabid beetle assemblages (Fig. 7.13.) as well as All Coleoptera (Fig. 7.14.) and Coleoptera (7.15.) assemblages generally varied with time since clearfelling. Clustering separated sites more or less according to their age and replicates of the same age generally formed pairs, although clustering indicated that the species composition of regrowth sites were often similar to some Old-growth sites. The pattern of similarity of species composition as determined by cluster analysis of All Coleoptera assemblages was very similar to that exhibited by carabids, with only two sites grouped differently (a Late regrowth site and an Old-growth site). The Mantel tests showed that there was a significant positive association between the UPGMA input matrices for carabids and All Coleoptera.

Other beetle taxa demonstrated more complex patterns of clustering. While same age regrowth sites often grouped together, there was no clear separation based on forest class and it was evident that old-growth sites were not characterized by a distinctly different fauna. Similarity in site groupings between carabids and other coleopteran taxon varied. Similarity in the patterns of site groupings was a little stronger between carabids and lucanids (Fig. 7.17.). The Mantel tests showed that there was a significant positive association between the UPGMA input matrices for carabids and Lucanidae. There was little similarity in the patterns of site groupings between carabids and Coleoptera; carabids and Curculionidae (Fig. 7.16.), carabids and Staphylinidae (Fig. 7.18.) or between carabids and tenebrionids (Fig. 7.19.). Pairwise comparisons of the site groupings exhibited by Coleoptera and other coleopteran taxon generally showed no patterns of similarity.

Weilangta Carabidae

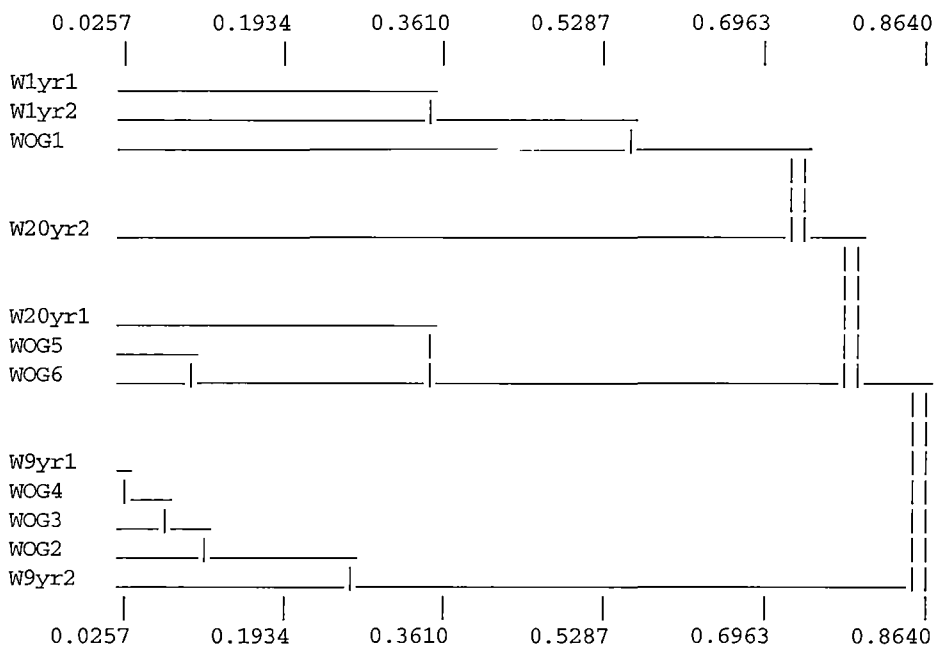


Figure 7.13. Dendrogram from UPGMA analysis, based on the species composition of the carabid fauna in the Weilangta forest, showing the degree of similarity between sites (|) within the four clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Details relating to the site codes used here are given in Chapter Four.

Weilangta All Coleoptera

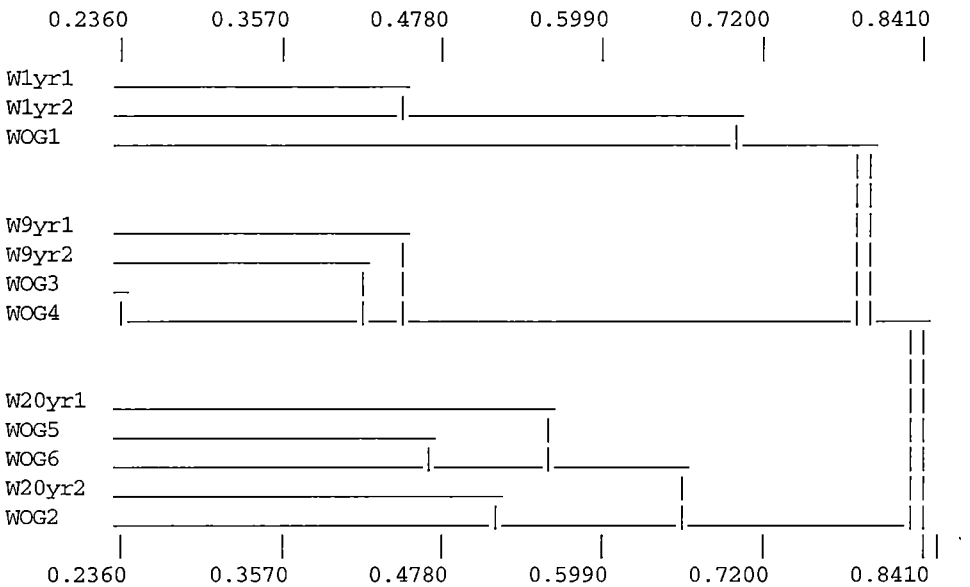


Figure 7.14. Dendrogram from UPGMA analysis, based on the species composition of the total ground dwelling coleopteran fauna in the Weilangta forest, showing the degree of similarity between sites (|) within the three clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Details relating to the site codes used here are given in Chapter Four.

Weilangta Coleoptera

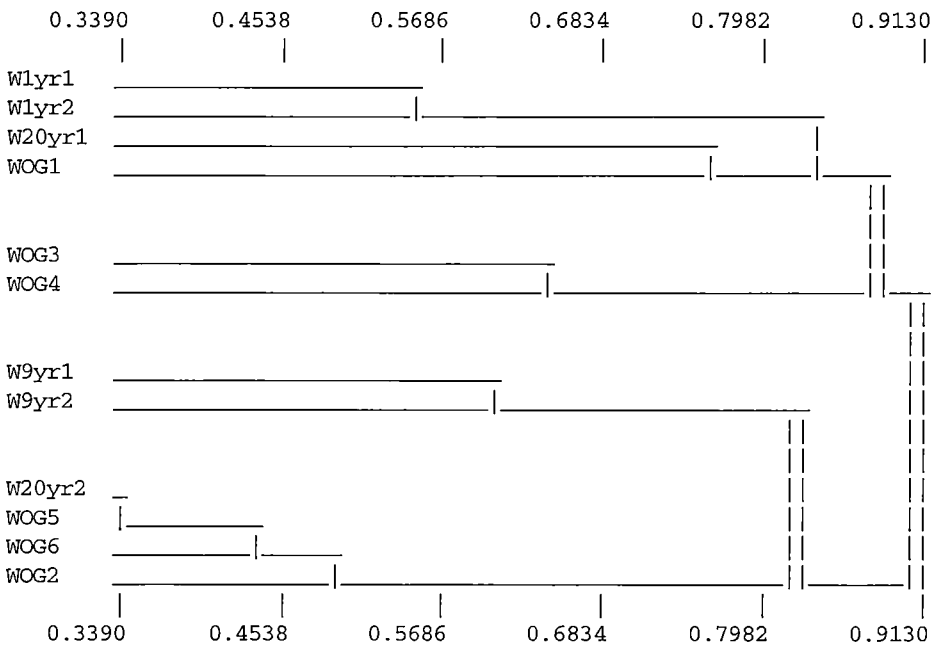


Figure 7.15. Dendrogram from UPGMA analysis, based on the species composition of the ground-dwelling Coleopteran fauna (excluding carabids) in the Weilangta forest, showing the degree of similarity between sites (|) within the four clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Details relating to the site codes used here are given in Chapter Four.

Weilangta Curculionidae

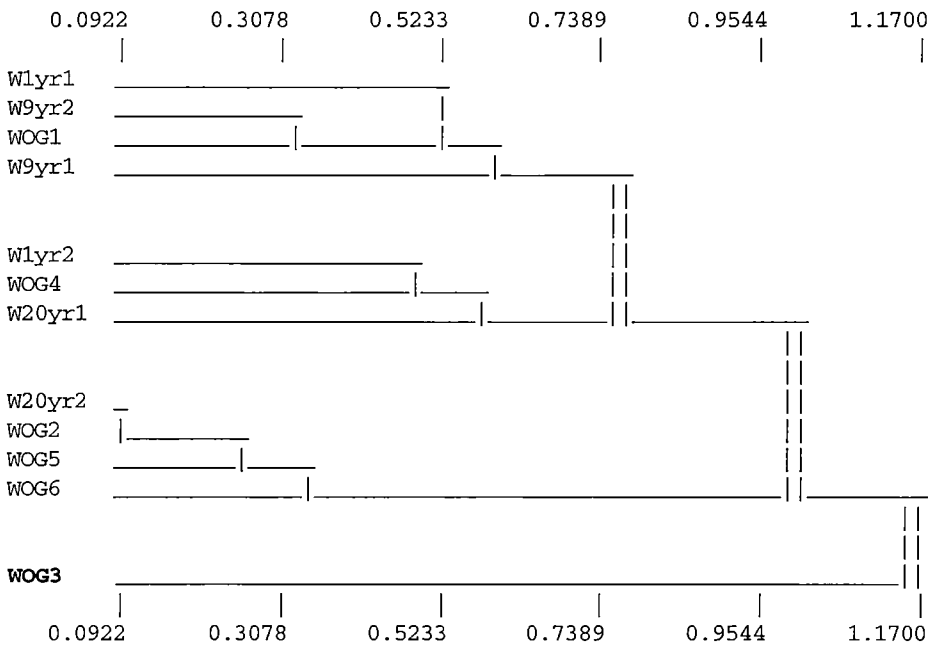


Figure 7.16. Dendrogram from UPGMA analysis, based on the species composition of the curculionid fauna in the Weilangta forest, showing the degree of similarity between sites (|) within the four clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Note: there were no curculionid species in the cluster with sites indicated in bold. Details relating to the site codes used here are given in Chapter Four.

Weilangta Lucanidae

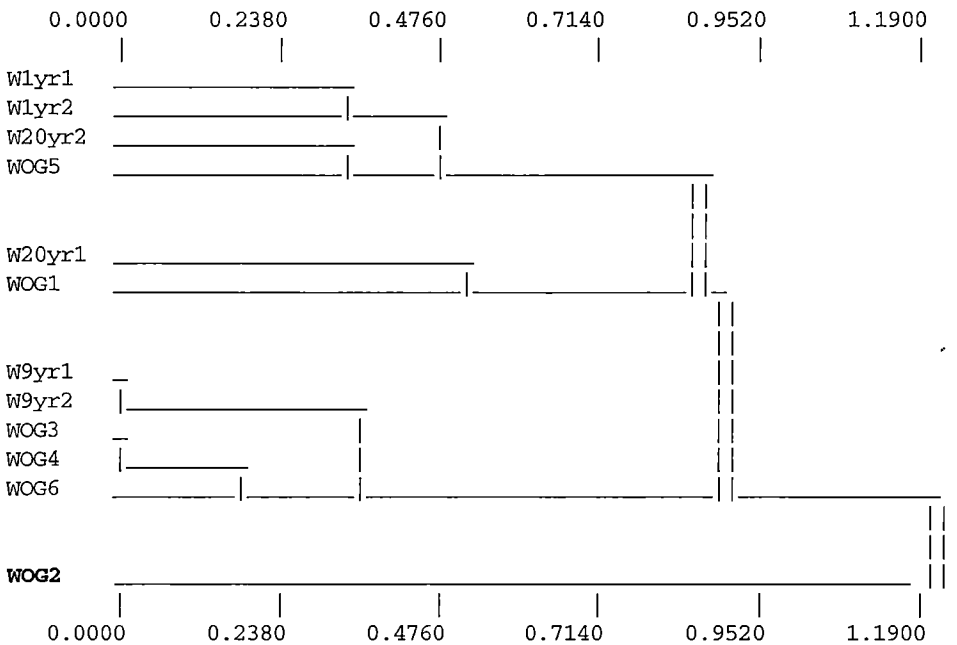


Figure 7.17. Dendrogram from UPGMA analysis, based on the species composition of the lucanid fauna in the Weilangta forest, showing the degree of similarity between sites (|) within the four clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Note: there were no lucanid species in the cluster with sites indicated in bold. Details relating to the site codes used here are given in Chapter Four.

Weilangta Staphylinidae

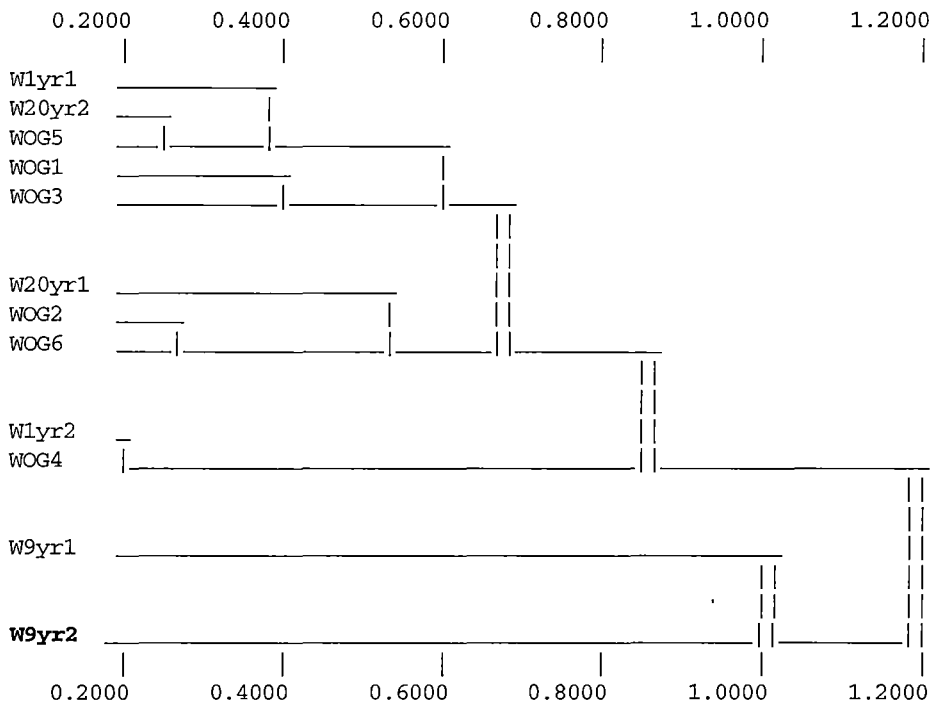


Figure 7.18. Dendrogram from UPGMA analysis, based on the species composition of the staphylinid fauna in the Weilangta forest, showing the degree of similarity between sites (|) within the five clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Note: there were no staphylinid species in the cluster with sites indicated in bold. Details relating to the site codes used here are given in Chapter Four.

Weilangta Tenebrionidae

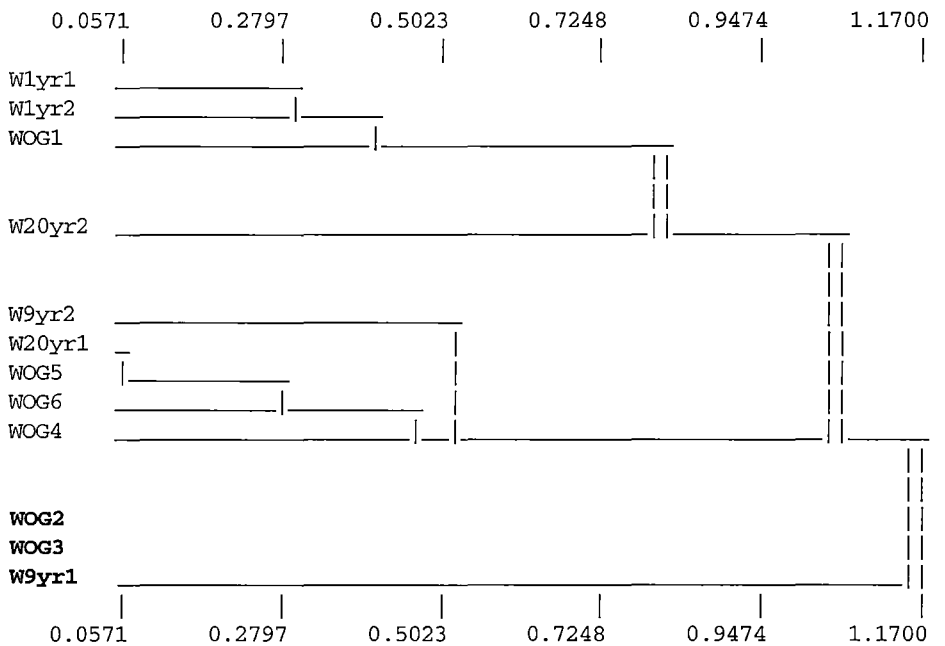


Figure 7.19. Dendrogram from UPGMA analysis, based on the species composition of the tenebrionid fauna in the Weilangta forest, showing the degree of similarity between sites (|) within the four clusters recognised by the PATN default setting, and the degree of dissimilarity between clusters (||). Note: there were no tenebrionid species in the cluster with sites indicated in bold. Details relating to the site codes used here are given in Chapter Four.

DISCUSSION

Taxon response to clearfelling

Three general types of response of the carabid fauna to large scale clearfelling of trees have been noted (Niemelä *et al.* 1993a; Niemelä *et al.* 1993b; Michaels and McQuillan 1995; Attlegrim *et al.* 1997; Niemelä 1997). (1) Some species (colonists) are restricted to or are most abundant in the Early clear-fell sites. (2) A few species (forest generalists) occur abundantly in most successional stages, and (3) primary forest (old-growth) specialists disappear or decrease in abundance in regrowth stands after logging.

In this study, the general response of all beetles to logging in this study fell into three groups. (1) Some species were more common or abundant in particular regrowth successional stages. (2) Generalist species were less sensitive to change and occurred abundantly in all successional stages (3) specialist species dependant on old-growth forest declined in abundance or disappeared from regrowth forests.

Carabid and overall coleopteran diversity was higher in the Early regrowth sites than in the regenerating or old-growth sites in both forests. This was also true for lucanids in both forest types and curculionids and tenebrionids in the Weilangta State Forest. Previous studies of the effects of logging on carabid beetles have found that an increase in species richness in early regeneration forest was due to the persistence of species associated with old-growth forest for some time after logging, combined with the invasion of new species into the disturbed area (Niemelä *et al.* 1993a; Niemelä *et al.* 1993b; Michaels and McQuillan 1995; Taylor *et al.* 1999). Colonisers were generally winged species (Michaels and McQuillan 1995), and/or species typical of open habitats elsewhere (Niemelä *et al.* 1993a; Niemelä *et al.* 1993b; Michaels and McQuillan 1995). The presence of the winged carabid *Mecyclothorax ambiguus*, a species common in sheep pastures (McQuillan and Ireson 1982) and native grasslands in Tasmania (Michaels 1997), in the Early regrowth sites in both forests supports this notion.

However, unlike carabids, there was little evidence that species of other beetles families occurring in the Early regrowth stages were species typical of open habitats elsewhere. A large proportion of species occurring only in the Early regrowth stages were flightless, and thus unlikely to be long distance colonisers. Various studies on the effects of clearfelling on other invertebrate groups (Huhta 1971;

Martikainen *et al.* 1996; Kaila *et al.* 1997; Niemelä 1997) on small mammals (Kirkland 1990; Mitchell *et al.* 1997) and on birds (Greenberg *et al.* 1995), have found that invading species may be species that prefer open forest conditions. For example, Kaila *et al.* (1997) found saproxylic beetle species associated with open forest habitats in clear- cuts and Martikainen *et al.* (1998) discovered several species of bark beetles associated with open forests living in logging residue in clear-cut areas. In this study, some species of lucanids persisted after the initial clear-felling and reached a peak in the intermediate successional stage for example, the saproxylic beetles *Lissotes curvicornis* in the Picton and *Lissotes sp. n.* in Weilangta. These may be species with a preference for the more open forest conditions created post tree-fall, that have colonised that log habitat still available post disturbance, or the newly recruited logs.

It is also likely that many of the flightless species occurring only in the Early regrowth were Old-growth species that had survived the logging. In Tasmania, forestry clear-fell operations typically leave large amounts of slash and stumps, which is subsequently burned. However, the burns are often patchy and incomplete and, depending on the intensity and the duration of the burn, many logs, extant and newly felled, survive, often relatively intact. Some individuals could potentially survive in or under logs and in the soil during the clearfell/burn. The importance of moist refuges, such as occur beneath logs and stones, for the survival of individual arthropods during and after regeneration burns, has been demonstrated by Madden *et al.* (1976) and Moldenke and Lattin (1990). Clearfelling may also have increased the likelihood of capture of some cryptic species, either because the removal of the vegetation forced sedentary species to become more mobile in search of a new suitable habitat, or because reduced ground cover and leaf litter increased trappability (Greenslade 1964; Melbourne *et al.* 1997). For example, the lucanids, *Lissotes politus* and *Lissotes subcaeruleus* were found only in the recently clearfelled sites in the Picton. Increased mobility due to the removal of habitat and ground cover would have increased the trappability of all surviving species. Many individuals trapped were therefore most likely associated with old-growth forest and reflect the species richness of the original and adjacent unlogged forest.

Curculionids, staphylinids and tenebrionids in both forests appeared to be colonists at a later stage in the succession, with several species being restricted to or more abundant in the Intermediate regrowth sites. Staphylinids are not very drought resistant and in forests many species are predominantly litter inhabiting species, as are tenebrionid species such as *Adelium abbreviatum*, and there is probably insufficient litter in Early regrowth forests. Bornemissza (1969) found that the

speed of reinvasion by invertebrates of burnt woodlands in Western Australia was associated with the development of leaf litter under trees. This may explain why community similarity between Early and Intermediate regrowth sites was greater than between Early and Late regrowth in the Picton, but the opposite in Weilangta, where there was greater species overlap between Early and Late regrowth. Regeneration time is shorter in wet sclerophyll than dry sclerophyll forest and this will influence the time needed for leaf litter build-up and consequently the length of time prior to recolonisation demonstrated by litter-inhabiting species.

Many species of staphylinids and tenebrionids appeared to only recolonise Late regrowth sites. Combined with the presence of forest generalist species, this resulted in Late regrowth sites in both forests having markedly higher species richness and abundance of staphylinids and tenebrionids. This probably reflects the attraction to moisture and decaying logs and other plant matter of some staphylinids and the dependence of many tenebrionid forest species on very wet and well rotted wood, and the fungi that grow on it, for food in the larval stage (Lawrence and Britton 1994).

Forest generalists were a marked proportion of all beetle taxa in the Picton, but a smaller proportion in Weilangta. Lucanids and staphylinids had no generalist species in Weilangta, but a markedly high proportion (two thirds) of their species were successional specialists. This could reflect differences in the plant communities. Dry and wet sclerophyll forests are ecologically distinct (Kirkpatrick and Bowman 1982). Dry sclerophyll forests are less dense than wet sclerophyll and lack tall, dense understoreys (Kirkpatrick and Bowman 1982) and regeneration time is slower. Particular microhabitats are therefore likely to be scarcer and more patchily distributed in regrowth stands in dry sclerophyll forest.

Many beetle species, particularly those defined as old-growth specialists, disappeared or decreased in abundance in regrowth stands after logging and some old-growth specialists were apparently absent from even the oldest regenerating stands. Studies on the impact of logging on carabids have also indicated that while many old-growth forest specialists appear to survive the immediate impacts of logging they are unable to persist in the longer term and are absent from older regeneration (Niemelä *et al.* 1993a; Niemelä *et al.* 1993b; Michaels and McQuillan 1995; Taylor *et al.* 1999). One possible explanation for this absence is the homogeneity of the forest environment at this stage (Niemelä *et al.* 1993a; Niemelä *et al.* 1993b; Michaels and McQuillan 1995; Niemelä 1997), with recruitment of woody debris to the ground suspended for some decades (Michaels and McQuillan

1995). The continuity of supply of decaying trees has been identified as an important factor for specialised saproxylic beetles (Siitonen 1994; Økland *et al.* 1996; Siitonen *et al.* 1996) and for mycetophylids (Økland 1996). Speight (1989) and Martikainen *et al.* 1998) found that a reduction in the amount of decaying wood in managed forests adversely effected a large number of saproxylic species.

Recent research demonstrates that the environment of old-growth forests is more heterogeneous, with a greater volume of sound and rotten downed logs (Andersson and Hytteborn 1991) than even near rotation-age regeneration forests (Franklin *et al.* 1981; Chandler 1986) or managed forests (Kirby *et al.* 1991; Reid *et al.* 1996). Larger quantities of wood increase the probability of wood occurring in all decay stages (Andersson and Hytteborn 1991). The even-aged forests created after clear-felling do not provide a continuum of trees and dead wood in different stages of decay and this will reduce the amount and species richness of wood-inhabiting fungi (Harvey *et al.* 1980).

Do different forest growth stages have characteristic beetle assemblages?

If age classes within a particular forest have characteristic assemblages, then samples of specific forest age classes would have similar community composition and the variation within an individual class should be less than that between different types.

The results of the Morisita-Horn community similarity indices demonstrate that carabid species composition showed less variation within grouped age classes (i.e. community composition was more similar between similar age sites) than between different age classes, in both forests. This was true for All Coleoptera and Coleoptera, but not necessarily for other beetle taxa considered separately. Lucanids, for example, showed less variation within grouped regrowth age classes than between different age classes, but species composition was extremely variable between Old-growth sites in both forests. Staphylinids and tenebrionids followed this pattern in the Picton, but in Weilangta, community composition was more variable within similar aged sites than between age classes; while curculionids followed this pattern in Weilangta, yet in the Picton, curculionid community composition was more variable within similar aged sites than between age classes.

The INDVAL index demonstrates quite clearly that different forest age classes have characteristic or indicator species, however, this does not mean that different forest

age classes will have discrete assemblages. While the variation in community composition within age classes may have been less than that between age classes for beetles generally, the Morisita-Horn community similarity Index values demonstrate clearly that there is overlap in community composition between age classes, the proportion of overlap varying with individual taxa. This is also evident from the UPGMA analysis, which shows considerable overlap in species composition between clusters and that age classes did not contain discrete assemblages. Similarly, two studies in western Canada focussing on whether carabid species specialise in use of old-growth forest habitats by Spence *et al.* 1996 found little evidence that the oldest forests (>120 years) contained unique assemblages of carabids.

The community composition at any given site will be made up of species restricted to that habitat type, including rare species, generalist species, some showing a marked preference for that habitat type, and a number of vagrants, transients or migrants (see also the discussion in Chapter Six). Habitat is the physical (biotic and abiotic) environment in which species and communities exist. Patterns of distribution and abundance will be determined by individual species specific responses to habitat (or to other individuals within the habitat). However, while habitat can vary markedly at many scales from centimetres to kilometres, individual species respond to their immediate surroundings and these responses can result in considerable variation in abundances within areas (such as regrowth class) that may be defined as a single habitat (Underwood and Chapman 1999).

In addition to stochastic variation in the location of subpopulations (Gilpin and Hanski 1991; Martikainen *et al.* 1998; Underwood and Chapman 1999), several factors may influence variability in species occurrence, particularly in old-growth stands. Extant assemblages of beetles are shaped by ecological events that change conditions locally. Old-growth forest in effect is a mosaic of successional stages and spatial heterogeneity within old-growth forests is maintained by mosaic processes. Mosaic processes such as tree decay or tree-falls or variation in litter quality will influence the availability of particular microhabitats in space and time (Haila *et al.* 1994). Similarly, beetle assemblages will be influenced by the extent to which successional stages mimics or reproduces preferred microhabitat conditions.

Suitable habitat for forest specialists is often patchily distributed and scattered even within natural forested landscapes (Chandler 1986, Niemelä *et al.* 1993a; Michaels and McQuillan 1995). For example, many lucanid species are thought to have extremely localised distribution within their overall range (Lawrence and Britton

1994), particularly those associated with wet forests (Wood *et al.* 1996, Meggs 1997) and species may not be present even in apparently optimum habitat ([Meggs, 1996a ; Meggs, 1997). Species associated with dead or decaying wood may have particular preferences for log and/or decay type (Ikeda 1987; Araya 1993a,b; Wood *et al.* 1996). Moreover, according to Martikainen *et al.* (1998), uncommon saproxylic species have stricter microhabitat requirements than other species and may inhabit only a few dead trees at one time within a forest.

Are carabids good ecological indicators

Ecological indicators are species (or species assemblage) who demonstrate a predictable and readily observable response to changes in environmental state and whose response reflects that of at least a subset of other biota. It has been demonstrated in this study and in an earlier study by Michaels and McQuillan (1995) that the response to logging demonstrated by carabids in Tasmania agrees with the response noted for carabids in Finland and Canada (Niemelä *et al.* 1993a; Niemelä *et al.* 1993b; Spence *et al.* 1996), and with the response noted for carabids in a range of forest types elsewhere (for example Sustek 1981; Lenski 1982; Jennings *et al.* 1986). There would appear to be clear evidence then that carabids meet the first criteria and demonstrate a predictable and readily observable response. The question is did their response reflect that of other Coleoptera? And, can they therefore be used to assess if general beetle diversity is changing in response to forest management?

Although carabids did not reflect the exact response of the individual beetle taxa considered separately, they did reflect the overall patterns of diversity exhibited by beetles as a group in response to forest management practices. Carabids therefore have demonstrated utility as ecological indicators of changes in overall beetle diversity and distribution in response to forest management.

Carabids as ecological indicators

Two main factors, fragmentation and isolation, have potentially adverse effects on the maintenance of diversity in logged forests (Soulé 1991; Spence *et al.* 1996; Niemelä 1997; Saunders *et al.* 1998). Clear-felling fragments continuous forest into a mosaic of clearcuts, regrowth and isolated stands of primary forest. While it can be expected that beetles adapted to mosaic processes within old-growth forest are also adapted to locating new suitable sites in response to small-scale

disturbances (Haila 1994), both the quantity and continuity of particular habitats is crucial.

Small forest fragments may not be able to support viable populations of all species, with rare and specialist species particularly at risk (Michaels and McQuillan 1995; Niemelä 1997; Martikainen *et al.* 1998). Biotype continuity is also a basic requirement for the maintenance of populations of species that depend on particular microhabitats associated with old-growth forest (Haila 1994; Michaels and Bornemissza 1999). These microhabitats need to be constantly reproduced within the dispersal radius of the species concerned. If the distance between fragments is too great for successful dispersal, and source areas are small and widely scattered in a highly fragmented forest, species with poor dispersal ability, such as flightless lucanids, may be unable to colonise suitable habitat (den Boer 1990; de Vries *et al.* 1996). Økland (1996) found the percentage of surrounding old-growth in the landscape to be a major factor for both species richness and population viability of mycetophylids.

Maintaining suitable habitat, sufficient source areas and biological connectedness between old-growth stands to enable dispersal has been identified as essential to maintain the integrity of the original forest fauna in managed forests (Taylor 1991; Haila 1994; Spence *et al.* 1996; Niemelä 1997; Michaels and Bornemissza 1999). In Tasmania, for example, forestry managers are committed to implementing these strategies. But only monitoring will establish whether management actions are in fact achieving the desired outcome of maintaining extant biodiversity at all scales. Thus for monitoring purposes in managed forests ecological indicators are crucial to indicate the success or otherwise of management practices in achieving these goals. Can carabids be used as ecological indicators to monitor for evidence of sustainability of specialist species and maintenance of source populations?

A set of beetle species were identified as specialist of old-growth forests, although the proportion varied with individual taxa. Several carabid species were restricted to or more abundant in old-growth, i.e. were old-growth specialists and this proportion was equal to or greater than that of the beetle community considered as a whole. Moreover, carabid old-growth species were flightless. Flightless forest species are generally considered to be poor dispersers, and thus, particularly susceptible to habitat disturbance and fragmentation (den Boer 1990; de Vries *et al.* 1996; Niemelä 1997; Michaels and Bornemissza 1999).

It could be expected therefore, that monitoring such species will provide evidence of the effects of fragmentation on species with poor dispersal powers and particular old-growth habitat requirements and that evidence of the maintenance of populations will reflect the success or otherwise of forest management activities for other old-growth beetles species. For example, the pileated woodpecker (*Drycopus pileatus* (L.)) was selected and is currently used as a Management Indicator Species by the USDA Forest Service because it is ecologically tied to old-growth mixed-conifer stands and the maintenance of pileated woodpecker populations is considered to indicate successful management for the retention of sufficient supply of old-growth for habitat (Torgersen and Bull 1995). Evidence that monitoring old-growth carabid species is likely to be successful is that other species identified as old-growth species were present in Late regrowth in Weilangta in similar proportions to carabid species defined as old-growth species while in the Picton the absence of carabid species defined as old-growth specialists in Late regrowth was matched by the absence of other old-growth beetles species.

Maximisation of regional habitat diversity has been suggested as a means of conserving biological diversity (Soulé 1991; Noss and Cooperrider 1994). In the forest context this will involve ensuring not only that all successional stages are included in the forest mosaic, but that interconnectivity is maintained. Results here suggest that carabids are useful indicators for studies on the effects of fragmentation. Support for this conclusion also come from a recent study in Tasmania, which used four of the carabid species identified by Michaels and McQuillan (1995) as Old-growth specialists *Chylnus ater*, *Notonomus politus*, *Stichonotus leai* and *Pterocyrtus* sp. A. (the first three also identified in this study), to evaluate whether strips of uncut forest, retained to maintain connectivity between larger areas of forest which will not be subject to timber harvesting, do provide suitable habitat for species of carabid beetles which are mature forest specialists (Taylor *et al.* 1999). If so, the strips could potentially function to ensure retained unlogged forest would not suffer from isolation effects (Taylor *et al.* 1999) . The general absence of the two most abundant species identified as mature forest specialists from the narrowest strips; *N. politus* from both and *C. ater* from one; was considered evidence that narrow strips may not cater adequately for old-growth dependent species (Taylor *et al.* 1999).

CONCLUSION

A discriminating test of sustainable forestry is the extent to which it meets the aim of conservation of biodiversity in native forests. Since it is not practical to monitor every species, we need ecological indicators to measure the impacts of our management strategies and assess the efficacy of our management practices for maintaining biological diversity. Indeed, indicators for sustainability of biological communities is mandated for (Commonwealth of Australia 1996).

The use of broad-scale indicators, such as the extent of age classes or successional classes within a forest type, have been suggested (Wales 1998). But using the extent of age classes or successional classes within a forest type as indicators assumes that different age classes of forest have distinct ecological and conservation values and a characteristic fauna (York 1999). Evidence here and elsewhere (Haila 1994; Niemelä 1997; York 1999) suggests that terrestrial invertebrates such as ground-dwelling Coleoptera respond most strongly to more localised habitat conditions. While it is important to maintain all successional stages in the forest mosaic, mere availability will not demonstrate maintenance of beetle biodiversity.

Diversity must be preserved at all scales (Noss and Harris 1986; Haila 1994; Niemelä 1997) and this requires that invertebrates such as beetles are incorporated into any monitoring system. Good indicators will enable us to predict the potential impacts of human activities on biological diversity across a hierarchy of spatial and temporal scales. The evidence here suggests that carabids are useful indicators to predict and monitor the effects of forest management activities on a wider range of ground-dwelling beetles.

However, ecological knowledge of different species roles in ecosystems is vital for long term conservation sustainability, we need to understand biodiversity patterns, the processes which alter these patterns and the consequences of biodiversity changes to ecosystem functioning. Biodiversity can not be understood nor successfully managed by compiling lists of species which do not take into account natural variability in the abundance and distribution of the species that comprise the diversity (Underwood and Chapman 1999). Assessments of the habitat requirements of beetles are central to understanding their distribution and abundance. While the use of indicators may provide a quicker, easier and cheaper (and at present only) solution to management decisions, it provides little understanding of the functioning of the system. Only long term ecological studies

will provide the information on factors influencing the abundance and distribution of species and their contributions to ecosystem processes.

Chapter Eight

Synthesis and Conclusion

Biodiversity is in crisis. Our ability to assess the quality of ecosystems and to identify key properties which can serve as indicators of ecosystem quality has become a major issue for conservationists. The driving force behind this is the need to conserve and sustain biodiversity in the face of accelerating habitat loss, destruction and degradation. But time is short, resources and expertise are scarce, it is just not possible to inventory or monitor all biota. The search for characteristic or indicator species for conservation, monitoring and management has thus become a major challenge.

Biodiversity indicators are taxa whose diversity reflects some measure of the diversity of other taxa in a habitat or set of habitats. I have demonstrated the utility of carabids as biodiversity indicators for overall coleopteran diversity. This thesis has shown that there is a strong positive correlation between patterns of diversity exhibited by carabids and overall coleopteran diversity and that protection for carabids will ensure proportional protection for other Coleoptera (defined in the context of this thesis as ground-dwelling Coleoptera), in a range of habitats and biomes in Tasmania.

Ecological indicators are taxa which demonstrate a predictable and observable response to environmental variations such as disturbance and whose response reflects that of at least a subset of other biota. I have demonstrated the utility of carabids as ecological indicators to predict and monitor the effects of forest management activities on a wider range of ground-dwelling beetles. This thesis has shown that carabids reflected the overall patterns of diversity and distribution exhibited by beetles as a group in response to forest management practices in two forest types in Tasmania.

CARABIDS AS BIODIVERSITY INDICATORS

To date, the search for biodiversity indicators has largely been unsuccessful. Other studies have shown little evidence of positive correlations between the species richness of a potential indicator and the richness of other groups for which data is

available. Areas rich in species of one taxon frequently do not coincide for different taxa, regardless of scale, and this message has been repeated across countries.

Flather *et al.* (1997) found no overlap in species rich sites between birds and butterflies at 25 sites where both taxa had been surveyed within Glacier National Park, Montana, in the United States. Oliver and Beattie (1996a) found no significant positive correlations between ant, beetle, and spider species richness from four forest types in Australia. Prendergast *et al.* (1993) found that the percentage overlap in the richest areas between all pairs of taxa (viz avians, butterflies, dragonflies, liverworts and aquatic plants), on a scale of 10 X 10 km sample grids within Britain, averaged only 15%, with the maximum overlap 34%, between butterflies and dragonflies. Similarly, in a study of a range of taxa, in South Africa, including many well-studied taxa that are frequently used as biodiversity indicators, (ie vascular plants, mammals and birds), van Jaarsveld *et al.* (1998) found that there was little concordance, with the highest overlap between birds and butterflies only 24%. Gaston and David (1994) found that only one or two of the hotspots of many of the numerous possible pairwise combinations of twelve higher taxa of animals and plants, on an equal area grid (squares of approximately 152,000 km) across Europe, were congruent. At a continental scale, Flather *et al.* (1997) noted that regions rich in mammals were not the same regions rich in birds, amphibians, birds or reptiles.

Although Pearson and Cassola (1992) recorded significant ($P<0.01$) positive correlations in species counts among tiger beetles, birds and butterflies across North America, Australia and the Indian sub-continent, Flather *et al.* (1997) found that much of the covariation in species counts could be accounted for by the general pattern of increasing diversity with decreasing latitude. Partial correlation analysis, used to remove the influence of latitude on species counts for all taxa, indicated much weaker patterns of association. Although a strong association between birds and butterflies remained, the correlation between cicindelid (tiger beetle) counts and butterfly counts was considerably weaker, and the correlation between cicindelid counts and bird counts was no longer different from 0 ($p = 0.20$). Moreover, while continental scale comparisons are valuable for comparing relative diversity of large areas, they have limited use at a scale relevant to appropriate jurisdictions and practical conservation planning, selecting and defining reserve boundaries (Flather *et al.* 1997).

In contrast, correlated species counts in this study, revealed a significant positive, and relatively strong covariation between carabid and All Coleoptera and between

carabid and other Coleoptera species counts in the three biomes investigated. The mean correlation among the three biomes was 0.57 and 0.51, (range 0.24 -0.82 and 0.41 - 0.67) respectively. While Flather *et al.* (1997) consider significant correlations of 0.5 or less show only weak concordance they still consider it evidence of covariation. Indeed, Pearson and Cassola (1992) identified tiger beetles as biodiversity indicators on the basis of a similar mean correlation of 0.54, range (0.36 - 0.73) between tiger beetle counts and bird counts.

Representativeness is a primary criterion in the selection of reserve areas, The Regional Forest Agreements (RFA) in Australia, for example (Commonwealth of Australia 1995). With regard to the indicator concept, the assumption is that areas, or sets of areas, that are species rich and represent the range of diversity for the indicator taxa will be species rich for other taxa. Although there have been few tests of potential indicator taxon using the representativeness criteria, the evidence suggests that there is unlikely to be accord between taxa on what constitutes a representative set. For example, Sætersdal *et al.* (1993) found that the six woods chosen to represent the maximum number of plants species were not the same six woods which represented the maximum number of bird species. Where twelve woods were chosen, only four overlapped with those chosen for birds.

In comparison, the Representative Species Richness approach, used in this study to determine a representative set of carabids, also ensured proportional representation for all Coleoptera. Unfortunately, one problem with the representativeness criteria is that it may not adequately conserve all species, since representative communities defined by classification often predicts only the presence of common and widespread species (Michaels 1999). For example, the most species rich wetlands from each of nine wetland types, while selected to ensure full representation of major wetland habitat types in northern New South Wales, included only 67% of all native wetland plant species (Margules *et al.* 1988). Similarly, in my study, the representative approach only conserved 67%-78% of all coleopteran species.

Representation is subtly different from the criterion of representativeness. Under the latter concept, sites are sought that represent archetypal communities, and once represented, a community type is considered protected. In contrast, representation means capturing the full spectrum of biological and environmental variation with the understanding that this variation is dynamic and not easily classified (Noss and Cooperrider 1994). There is now abundant evidence that, where the goal of site selection is to represent the greatest numbers of taxa in the least area, or an agreed quantum, algorithms which take into account the complementarity of biota between

sites are far more efficient than those based solely on relative richness (Kirkpatrick 1983; Margules *et al.* 1988; Vanewright *et al.* 1991; Pressey *et al.* 1993; Williams *et al.* 1996; Balmford *et al.* 1996a). An effective biodiversity indicator taxon in the context of complementarity would therefore be one for which a set of areas ensuring representation for the indicator group will be species rich in general (Faith and Walker 1996b).

To date, available evidence has suggested that complementary sites for one taxon rarely coincide with those for other taxa. Using plant and bird lists from 60 deciduous woods in Norway, Sætersdal *et al.* (1993) found that the minimum set of 32 areas representing all plant species shared only 5 sites with the minimum set of 12 areas with complete representation of birds. Similarly, pairwise comparisons of complementary sets for vascular plants, birds, butterflies, mammals, termites, antlions and buprestid beetles and scarabaeioid beetles revealed a mean overlap of 10% with a maximum overlap of 21% between buprestids and butterflies (van Jaarsveld *et al.* 1998). In contrast, the results of the complementarity approach here, demonstrate that a set of areas species rich for carabids are also species rich in general for all Coleoptera, since representation for all carabids gave protection to around 90% of all coleopteran species.

My results support the current idea that correlations between taxa are most likely where differences in lifeways are reduced (Hammond 1994). For example, Oliver and Beattie (1996b) found that richness within the family Carabidae was significantly correlated with richness within all other beetle families (Pearson $r = 0.61$, $n = 20$, $P < 0.01$) in four Australian forests sampled. Similarly, Beccaloni and Gaston (1995) found that the diversity of the butterfly subfamily, Ithomiinae, correlated well with total butterfly diversity. It seems likely therefore, that we will have to focus the search for biodiversity indicator taxon at the order level rather than searching for the unattainable holy grail of one single all encompassing higher taxon. For example, Hunt (1994) suggests that within the order of orabatid mites particular families which are more 'user friendly' i.e. more readily recognisable may be selected as 'priority taxa' to circumvent difficulties with identification and numbers of individuals.

CARABIDS AS ECOLOGICAL INDICATORS

The results in this thesis showed that carabids demonstrated an observable and measurable response to forest management practices. Moreover, my results show that this response agrees with the response noted for carabids in a range of forest

types elsewhere, such as spruce-fir forest in Maine (Jennings *et al.* 1986), pine forests in Czechoslovakia (Sustek 1981), oak forests in the U.S. (Lenski 1982), and boreal forest in Finland (Niemelä *et al.* 1988) and Canada (Niemelä 1993b; Spence *et al.* 1996). This is clear evidence that carabids meet the first criterion for an ecological indicator and demonstrate a predictable and readily observable response. Further, it shows that carabids indicate a specific ecological pattern at the desired spatial and temporal scale which makes them a suitable ecological indicator group (Kremen 1992).

In addition, the response demonstrated by the family Carabidae reflected the overall patterns of response demonstrated by beetles as a group, although it did not necessarily reflect the exact responses noted for other beetle families considered separately. The main difference being that individual species of some families appeared not to colonise early regrowth sites but tended to be colonisers of later successional stages, dependent on the extent to which successional stages reproduced their preferred microhabitat conditions. Although, it is possible that species from these families were not recognised as colonists because of the general lack of knowledge available. This study defined those species occurring only in the early regrowth stage colonists if winged, and old-growth species who had survived the initial clearfell, if flightless. Only further studies will enable us to accurately classify all forest species.

It is difficult to compare these results to other studies, since published studies demonstrating that the response of a proposed ecological indicator reflects that of at least a subset of other biota could not be found. For example, in Canada, the Ontario Crown Forest Sustainability Act (CFSA) passed in 1995 requires Ontario to demonstrate that forest harvesting is conducted in a manner ensuring sustainability of forest ecosystems (McLaren *et al.* 1998). The selection of indicators of forest sustainability to monitor and assess the results of management is a key part of the new management regime. Species were selected as indicators of biological diversity with the assumption that they are correlates to various elements of biodiversity (Thompson and Angelstam 1998; McLaren *et al.* 1998) and as ecological indicators with the implicit assumption that their response to forest management reflects that of at least a subset of other biota (at least biota within their home range sizes) (McLaren *et al.* 1998). While such studies provide critical information on the target taxa (the group under investigation or the object of attention (Hammond 1994), it has yet to be demonstrated that the responses exhibited reflect the responses of other related or unrelated taxa within the ecosystem or habitat.

Implications for forest management

In the forestry context, the objective of monitoring ecological indicators is to determine whether or not forest harvesting is having long- term impacts on wildlife populations and their habitats (Mclaren *et al.* 1998). The species of main concern are old-growth specialists associated with microhabitats of old-growth forest, such as coarse woody debris and fallen and decaying logs (Niemelä 1997; Michaels and Bornemissza 1999), particularly saproxylic species (Kaila *et al.* 1997; Martikainen *et al.* 1998; Michaels and Bornemissza 1999). Although carabids generally are not saproxylic, decaying woody material and leaf litter are crucial elements of habitat for many forest-dwelling beetles species (Niemelä *et al.* 1992; Haila *et al.* 1994; Spence *et al.* 1996).

Fragmentation and edge effects are two processes thought to have adverse effects on such populations. Two studies in western Canada by Spence *et al.* (1996) demonstrated that carabid assemblages will be subject to edge effects resulting from patchwork logging on the landscape. Such effects will be indicated through colonisation by open habitat specialists. Where patches of uncut forest are too small to maintain viable populations of old-growth specialists, they will be characterized by habitat generalists. Since the response of carabids reflects the response of ground-dwelling beetles as a group, it can be expected that monitoring carabids can provide useful evidence of the effects of fragmentation and edge effects on other old-growth beetles.

The results of my thesis and of other studies suggest that, in the monitoring context, species richness is likely to be a poor criterion to judge the impact of forest management on wildlife populations. While fragmenting primary forest with clear-cuts may increase species richness at a local scale, it would not contribute to species richness at a larger scale if sensitive species were lost from the landscape (Niemelä *et al.* 1993a; Noss and Cooperrider 1994; Michaels and McQuillan 1995; Niemelä *et al.* 1996; Spence *et al.* 1996; Michaels and Bornemissza 1999). Species richness tells us little about changes in species composition. Only monitoring of recognisable species will tell us how successful management practices are. For example, in study of fragmentation effects, Davies and Margules (1998) found no significant differences in carabid species richness although there were distinct changes in species composition. Similarly, no difference in species richness or H' (Shannon's diversity index) were found for ants or birds in a range of rangeland sites covering a gradient from irreversibly degraded to healthy sites and including sites exposed to grazing stress (Whitford *et al.* 1998). They concluded that

measures other than species richness or species diversity indices need to be used as indicators of biodiversity for rangeland health assessments.

Concern is growing world-wide about the loss of old-growth forests as habitats (Niemelä *et al* 1993b; Noss and Cooperrider 1994; Spence *et al.* 1996). While many believe that the solution is to save someone else's rainforests, forests are being exploited wherever they occur (Spence *et al.* 1996). If we are to predict and modify the effects of forestry practices on obligate forest species and ensure that forests are harvested according to principles of ecologically sustainable management, we need ecological indicators to measure the impacts of our management strategies and point out how well we are doing at maintaining biological diversity (Vora 1997). But, if we are to make management decisions on the basis of a sample of the biota, there must be some confidence that the sample can be extrapolated to the larger proportion of species that have not been sampled. My results show that the response of carabids to forestry practices can confidently be extrapolated to a wider range of ground-dwelling forest beetles.

CARABIDS AS INDICATORS

A range of both desirable and necessary characteristics of indicators have been repeatedly suggested. It has now been demonstrated that carabids fit all criteria. In Australia, as elsewhere, carabids are widespread, present in almost all kinds of habitats, generally in reasonable numbers, all the year round. They demonstrate strong affinity to particular habitats are responsive to habitat change and changes in land management and are readily and cost-effectively sampled with modest effort and without the use of specialised equipment using standardised and tested trapping techniques. Relatively readily identifiable, they are one of the most widely studied taxa in the world. This thesis has shown that carabids also reflect the response of at least a subset of the biota, i.e., a wider range of ground-dwelling Coleoptera.

My thesis and other studies in the Australian context (Michaels and McQuillan 1995; Michaels and Mendel 1998) show that carabids here exhibit similar ecological patterns and complexity to carabids elsewhere. They are indeed an ecologically informative group. Moreover, in providing for their needs and for using animals to monitor environmental health, it is unrealistic to ignore one of the most diverse and sensitive taxa in species assemblages (New 1995). In addition, it is unlikely that invertebrates such as beetles will be adequately conserved under the umbrella taxa of plants or vertebrates (Yen 1987; Michaels and Mendel 1998). Conservation assessment and monitoring must include representative invertebrate groups. Since

the greatest returns to practical conservation will be gained by determining patterns of diversity and distribution of selected, ecologically informative invertebrate groups and using those patterns in helping to set priorities for habitat conservation and effective protection of natural assemblages (New 1995), it would be foolish to ignore the potential of carabids for this purpose in the Australian context.

Unfortunately, as pointed out by New (1998), the major difficulty in employing carabid beetles as (biodiversity and ecological) indicators in the Australian context is that they have so far received little attention and as a result, our knowledge of the biology and distribution of most Australian species is scant. Plus, good user friendly field guides and keys are generally lacking. However, this problem is not restricted to carabids. Our knowledge of the taxonomy and distribution of most invertebrates in Australia is poor, particularly at the species level (Vane-Wright *et al.* 1994; Oliver and Beattie 1993; Oliver and Beattie 1996a,b; New 1995; Cranston and Trueman 1997). User-friendly taxonomic and diagnostic keys to species level are scarce for all invertebrate taxa, and although the prime aim of ABRS (Australian Biological Resources Study), the major source of funding for taxonomic studies, is to provide the taxonomic tools needed to document and manage Australia's biota, funding for identification aids to species level for invertebrate taxa is meagre.

Collections are the physical database and the underlying knowledge base for the investigation of the diversity of Australia's biota. As repositories for named organisms that authenticate identification (Vane-Wright and Cranston 1992),they represent samples of variation, distribution, seasonal and annual occurrence of organisms. For many species there is no other information of presence, distribution and biology than that contained in collections, for example, 50% of invertebrates known to occur in Australia have not been recorded in the literature (are not named) and are known only from collections (West and Nielson 1992). Collections are the most cost-effective record of the Earth's biological diversity (Raven 1992). Yet the size of the workforce employed to carry out taxonomy and care for all the collections in Australia, has steadily decreased since 1975 (Richardson and McKenzie 1992).

Fortunately, Australia has a reasonable number of carabid species (around two thousand species), and, unlike say ants with many of the 4000 or so species unnamed (New 1998), the great majority of Australian carabids have been named (New 1998). This is of great importance, because conservation is based at the species level. Utilising the valuable information on species distributions currently

available in various collections and collating and databasing all available information, such as data on museum specimens, published and unpublished 'collector and scientific intelligence', such as was done for the British butterflies (New 1995), we would have a much better understanding of Australian carabids and the factors influencing their distribution and abundance and how they may be effected by anthropogenic factors such as land use or vegetation changes. Furthermore, it would take comparatively little time or resources to (a) provide an interactive CD ROM as a key to all species, and (b) map this information to provide information on distribution and correlate it with available information on habitat/vegetation/soil type etc. using Geographical Information Systems (GIS) for example. Remote sensing has been suggested as a valuable tool in support of biodiversity research (Soulé and Kohm 1989), and Davis *et al.* 1990) and Stoms and Estes (1993) propose an approach to integrate existing data on species distribution and habitat characteristics in biodiversity assessments using GIS technology, supported by remote sensing inputs. In Tasmania, Government conservation agencies are making increasing use of this technology for this purpose.

Carabid data could be incorporated into this system with little difficulty. The vegetation community type classification concept could readily be revised to associate community types to respective habitat types. For example, rigorous investigations on the relationships between vegetation types and the distribution and abundance of fauna will provide the information needed to relate vegetation classifications to fauna/wildlife habitat types (Pressey 1994; O'Neil *et al.* 1995; Buckton and Ormerod 1997). Occurrence of each carabid species can be listed in relation to "habitat type", and used to provide baseline data for ecologically based information storage-retrieval systems for land management, the Interim Biogeographic Regionalisation for Australia (IBRA) for example. IBRA represents a landscape-based approach to classification and is an integrated classification of biotic and abiotic variation, intended to define, map and describe the major ecosystems of Australia and (Saunders *et al.* 1998).

Classification techniques can be used to identify and group the ecological elements which influence carabid distribution (Michaels 1999), while the Indicator Value (INDVAL) method used here to determine indicator species for forest successional stages can be used to identify indicator species corresponding to the various habitat/plant community types. The INDVAL approach can be generalised to any classification of sites, based on either species distributions or on a priori ecological variables or land classifications (Dufrene and Legendre 1997). Species,

environmental variables and habitat types can be input into schemes such as (Tasmanian Parks and Wildlife Services (TASPAWS) data base in Tasmania. GIS can be used to map and predict habitat and this can be followed by ground truthing. For example, GIS has been used recently to predict richness of all vertebrates from their individual vegetation cover type preferences (Scott *et al.* 1993). Lists of the species assemblages characteristic or typical of that plant community type (or ecological factor combinations) can then be used to evaluate the comparative richness of sites, or the effects of isolation or fragmentation (Dufrene and Legendre 1997). Moreover, the coincidence of particular vegetation types or plant species or soil types with particular kinds of terrestrial invertebrates may give valuable leads to understanding the factors influencing natural distribution, or to identify gaps (New 1995).

CONCLUSION

While established evidence suggests that we can only truly conserve what we know and understand, practical realities based on economics mean we cannot inventory and monitor all species, so we have no option but to use surrogates or indicators. Conservation efforts for invertebrates such as beetles are daunting because of the sheer weight of numbers. But strategies that fail to include beetles and other invertebrates are likely to fail to conserve this, the major portion of all biodiversity and in the process may fail to maintain essential ecosystem processes. If we must use surrogates or indicators then we must do so at all relevant scales. The taxa in any 'shopping basket' must include representatives of invertebrate groups such as beetles.

Australia along with 156 other countries is signatory to the (1992) Convention on Biological diversity (Australia 1996). The convention calls upon its member states to identify components of biological diversity important for the long-term conservation and sustainability of biodiversity such as indicator species (McFadgen and Simpson 1994). It also calls upon signatories to monitor the components of biological diversity and identify activities which are likely to have significant adverse impacts. In order to meet its international obligations, Australia intends to develop a "core set "of biodiversity indicators to assess and monitor the condition of Australia's biological diversity (Saunders *et al.* 1998).

The results reported in my thesis support the hypothesis that the family of ground beetles (Carabidae) is an appropriate biodiversity indicator taxon for identifying and predicting the biodiversity patterns of ground dwelling Coleoptera in most but not

all instances in Tasmania. Results also support the hypothesis that carabids are useful ecological indicators to predict and monitor the effects of forest management activities on a wider range of ground-dwelling beetles. Carabids are therefore an appropriate group to incorporate into this "core set".

Strategies for biodiversity conservation include both short term actions and long term programmes. In the short term, we may have to rely on indicators, but in the long term it is absolutely critical that we put resources into increasing the state of knowledge of biodiversity. But, it is equally critical that we put resources into indicators in the short term. Lack of funding underlies the taxonomic impediment. Demand for taxonomic expertise is increasing, yet the number of taxonomists and financial resources for taxonomic research is declining and the shortage of taxonomic expertise is especially acute in invertebrate conservation, primarily due to policy failure and consequently a lack of funding. Moreover, in spite of the rhetoric, biodiversity studies are still woefully underfunded. It is totally unrealistic to expect that indicators will be an adequate solution to the biodiversity crisis if they are inadequately funded.

Chapter Nine

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